# Greenhouse gas emissions and potential mitigation options for the Australian dairy industry

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Submission in fulfilment of the requirements for the degree of Doctor of Philosophy

University of Tasmania

March 2019

## Declaration of Originality

The publications presented for the degree are the original works, written in conjunction with others. In all publications, the candidate was the senior author. The estimated percentage contribution of the candidate is shown in the list of publications submitted for the degree under the 'Statement of Co-Authorship' section of the thesis.

This thesis contains no material which has been accepted for a degree or diploma by the University or any other institution, except by way of background information and duly acknowledged in the thesis, and to the best of my knowledge and belief no material previously published or written by another person except where due acknowledgement is made in the text of the thesis, nor does the thesis contain any material that infringes copyright.

Signed: Date: 5th March 2019

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## Abstract

One of the biggest challenges facing the world today is how we feed an ever-increasing population while reducing greenhouse gas (GHG) emissions that are contributing to global warming. Unquestionably, the livestock sector represents a significant source of emissions, generating carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$).

In 1988, the Intergovernmental Panel on Climate Change (IPCC) was established to prepare a comprehensive review and recommendations concerning to the state of the science of climate change, the social and economic impact of climate change, and possible response strategies and elements for inclusion in a possible future international convention on climate. The first assessment report of the IPCC served as the basis for negotiating the United Nations Framework Convention on Climate Change (UNFCCC). Via the IPCC, a series of algorithms and emission factors (EFs) were developed to calculate GHG emissions that conform to the UNFCCC, thus allowing individual countries to calculate their GHG emissions.

The Australian Federal Government began accounting and reporting the nation’s GHG emissions in 1990 according to the UNFCCC rules. Currently, agriculture is responsible for 13% of Australia’s GHG emissions and is the primary source of $CH_4$ and $N_2O$ emissions. The national accounting of GHG emissions adopts a large-scale approach. As such, it does not estimate individual farm GHG emission profiles, nor identify potential mitigation strategies to reduce total farm emissions.

The purpose of this thesis was to determine the GHG emissions of Australian dairy farms using the Australian GHG methodology and examine potential mitigation options to reduce on-farm GHG emissions attributed to milk production. To ascertain GHG emissions, a localised focus within one region was explored, where the milking herd grazed pastures year-round with supplementary feeding occurring either in the dairy parlour or grazed paddocks. Sixty dairy farms in Tasmania were examined, with their total GHG emissions varying between 704 and 5,839 t $CO_2$ equivalents ($CO_2e$)/annum. A metric of emissions intensity (EI) of milk production, defined as kg $CO_2e$/kg fat and protein-corrected milk (FPCM), was calculated to allow comparison between farms. The mean EI was 1.04 kg $CO_2e$/kg FPCM, with individual farms varying between 0.83 and 1.39 kg $CO_2e$/kg FPCM. Linear

regression analysis showed that 93% of the difference in total farm GHG emissions could be explained by annual milk production. The study also found that 60% of the difference in the EI of milk production between farms was explained by differences in feed conversion efficiency (FCE; kg FPCM/kg dry matter intake (DMI)) and nitrogen (N) fertiliser application rates (kg N/ha.annum).

This on-farm evaluation at a local level (Tasmania) was expanded nationally to 41 Australian dairy farms. Farms varied between grazing pastures with supplementary feed delivered in the dairy parlour and paddocks through to farms where, in addition to grazing and supplements delivered in the dairy, cows spent a proportion of their time off paddock consuming partial mixed rations. Individual farm total GHG emissions varied between 411 and 9,416 t $CO_2$e/annum. The Australia-wide mean was estimated as 1.04 kg $CO_2$e/kg FPCM, with individual farms varying between 0.76 and 1.68 kg $CO_2$e/kg FPCM. Linear regression analysis showed that 95% of the difference in total farm GHG emissions was explained by annual milk production. Milk production per cow (kg FPCM/cow.lactation) explained 70% of the difference in EI between farms. Farms were grouped according to their farming system (FS), indicative of the level of grain feeding and supplementary feeding management. The mean EI of milk production for FS1 farms (refers to grain feeding of < 1 t dry matter (DM)/cow.lactation) was 1.23 kg $CO_2$e/kg FPCM. This was significantly ($P < 0.05$) greater than the mean EI for FS2 and FS3 farms, at 0.98 and 0.97 kg $CO_2$e/kg FPCM, respectively. FS2 farms refers to grain feeding of > 1 t DM/cow.lactation with supplementary forage fed in the paddock while FS3 farms refers to grain feeding of > 1 t DM/cow.lactation and the incorporation of supplementary feeding into a partial mixed ration delivered on a feedpad.

The Australian inventory methodology for calculating GHG emissions is an estimation method based on current science. As new science pertaining to GHG emissions emerges, the inventory is updated, with the most recent occurring in 2015. An assessment was undertaken to ascertain the consequence of the updated methodology on the EI of milk production utilising the same 41 Australian dairy farm case studies. Mean EI increased by 3% to 1.07 kg $CO_2$e/kg FPCM (ranged between 0.84 and 1.54 kg $CO_2$e/kg FPCM). Annual milk production remained a strong determinant, with 96% of total farm GHG emissions explained by this. A Concordance Correlation Coefficient analysis was undertaken to estimate the extent of agreement between the two methodologies. There was moderate agreement between methodologies for estimating individual farm EI of milk production. However, primarily due to a regional variation in an emission factor (EF) for manure management, there was poor agreement between methodologies for estimating regional EIs. This study reaffirmed that while enteric $CH_4$ emissions remains the largest component of on-farm GHG emissions, waste $CH_4$ emissions has emerged as a more substantial source of on-farm GHG emissions.

The need to identify mitigation options that are considered ‘win:win’ options in reducing the on-farm GHG emissions while maintaining or improving productivity and/or profitability are critical to meeting the need to reduce ruminant livestock GHG emissions. In addition, win:win strategies may be more readily implemented by farmers, as opposed to mitigation strategies that erode productivity or profitability, in the pursuit of reducing GHG emissions. This thesis explored the GHG emissions reduction potential of two mitigation options applicable for Australian dairy farms;

(i) evaluating dietary and breeding approaches for improving animal N use efficiency (NUE);

(ii) improving feed quality to increase liveweight gain (LWG) promoting earlier mating of dairy heifers.

Reducing the overall diet N concentration was found to be a more effective means to improving NUE and reduce $N_2O$ losses than increasing the concentration of N in milk of lactating cows when modelled across three climatic regions. Nitrous oxide emissions were reduced by 50 to 57% when the supplementary feed was reduced from 4% to 1% N (total diet N concentration of 4.1 and 2.5%, respectively). In contrast, when the N concentration of milk was increased from 0.50 to 0.65%, reflecting 3.1% and 4.1% milk protein, $N_2O$ emissions were only reduced by 7 to 11%. This was an important finding, highlighting that reducing the source of N intake in the diet resulted in a more significant reduction in emissions compared to increasing the sink of N into milk. In addition, manipulation of dietary N would be a much easier mitigation option to implement than manipulation of N concentration in milk through breeding. This is a currently available mitigation option for all Australian dairy farmers to consider implementing, especially for those farms currently feeding a high protein diet.

Low quality diets, such as found in the subtropics, has resulted in a delay in heifers reaching mating liveweight (LW) to calve at two years of age. Improving the feed quality of the heifer diet from 9.5 to 10.9 megajoules of metabolisable energy/kg DM reduced the time heifers required to reach target LW for mating by 5 to 7 months. Enteric $CH_4$ emissions over the period between weaning and mating were reduced by 0.4 to 0.5 t $CO_2$/head. Collectively, increasing LWG, lowering enteric $CH_4$ emissions and earlier calving all contribute to reducing lifetime total emissions and thus the EI of milk production.

A Marginal Abatement Cost Curve (MACC) analysis was also undertaken, examining seven currently available mitigation options, across four representative Australian dairy farms, to ascertain the profitability of implementing mitigation options that reduce GHG emissions. The EI of milk production declined for all four farms across all seven mitigations (with only one exception for one farm when implementing one of the mitigation options explored), varying by between 0.01 and 0.05 kg $CO_2$e/kg FPCM depending on the farm and the mitigation option examined. When this decrease in EI was multiplied by the corresponding milk production, the decline in total farm GHG emissions varied between 5 and 233 t $CO_2$e/annum. The cost-effectiveness of each mitigation option, defined as the cost of implementation divided by the amount of $CO_2$e abated, found three of the seven mitigation options were profitable across all four farms, thus a triple win in terms of reducing emissions while increasing farm productivity and profitability. In addition, one other mitigation option was potentially profitable, but for only one of the four farms examined. The MACC analysis highlights the importance of reviewing inefficiencies on farm to determine which combination of mitigation options delivers the greatest reduction in total farm GHG emissions.

Benchmarking the GHG emissions of the Australian dairy industry has established that the industry EI average is comparative to those from other developed world nations. The breadth of variation in EI across the farms examined in this thesis has illustrated that there is scope for reducing the EI of milk production, both on individual farms and as a national average. While all seven mitigation options examined in the MACC analysis delivered a reduction in EI, six of these mitigation options also delivered a net reduction in GHG emissions. This is an imperative characteristic for the uptake required to lower GHG emissions from food production as part of the movement towards global carbon neutrality. While there is currently no pressure on individual dairy farms, or the national industry at large, to reduce their EI of milk production, global pressure and policy is progressing towards monitoring and reporting EI of milk production, with an increased focus on reducing net emissions. Farmers will need to adopt a range of additive and complementary mitigation options that reduce net emissions, with the residual, unavoidable emissions offset through other avenues such as sequestration.

***Impact statement***

This thesis contains five published research papers from the last eight years that have been drawn on by Australian dairy industry representatives and policy makers in the development of recommendations to reduce on-farm GHG emissions. Examples of this include a combined 66 journal article, book chapter and conference proceedings citations. The incorporation of industry averages from Chapter 4, and subsequent results from Chapter 7, have allowed for effective benchmarking of GHG emissions for the Australian dairy industry, through a range of avenues, including the on-line Dairy Australia DairyBase tool. Other examples include contributions to the Australia Dairy Industry Sustainability Framework and incorporation of benchmark results in a series of videos developed by Dairy Australia focusing on sources of on-farm GHG emission and mitigation options to reduce the carbon footprint of the Australian dairy industry.

## Acknowledgements

*"Do the difficult things while they are easy*

*and do the great things while they are small.*

*A journey of a thousand miles must begin with a single step"*

(Lao Tzu, Chinese philosopher, 6th century BC).

When I published my first 1st-author paper, back in 2011, I never envisaged that this would be the first step towards attaining a PhD. Researching greenhouse gas (GHG) emissions of Tasmanian dairy farms had never been done previously, was novel work that interested me and fitted into my work program at the time. Several papers followed on from this, all contributing to the projects I was employed on at the University of Tasmania. It was not until late 2017, that the concept of collating these papers together into a story, that has become a PhD, became visible. To assume that this PhD has been easy, because I have completed this using already published papers, would be incorrect. While I have not had to endure the pressures of three years of not knowing if my research was of publishable standard, the research question, by its very nature, has required me to consider the wider implications of GHG emissions, not only within the Australian dairy industry, but also the broader global context. Global GHG emissions requires a global plan, with my research just one component of many parts of the complexity.

I would like to thank a number of people for the contribution and support in the preparation of this thesis. Firstly, I acknowledge the contribution of all the co-authors of the publications presented in this PhD. Without these papers, there would be no PhD by Prior Publication;

- Ass. Prof. Richard Rawnsley, Tasmanian Institute of Agriculture, University of Tasmania, Burnie, Tasmania;
- Prof. Richard Eckard, Faculty of Veterinary and Animal Science, University of Melbourne, Parkville, Victoria;
- Dr Cameron Gourley, Victorian Department of Primary Industries, Ellinbank, Victoria;

- Mr Ivor Awty, Victorian Department of Primary Industries, Ellinbank, Victoria (retired);
- Dr Matthew Harrison, Tasmanian Institute of Agriculture, University of Tasmania, Burnie, Tasmania;
- Dr Leigh Trevaskis, Trinity College Queensland, Auchenflower, Queensland (previous address during publication was Emmanuel College, University of Queensland, St Lucia, Queensland);
- Mrs Catherine Phelps, Dairy Australia, Southbank, Victoria

I wish to thank Assoc. Prof. Richard Rawnsley for encouraging me to present my collection of publications in a form for recognition of a PhD by Prior Publication thesis, and for supervision of this thesis, along with Dr Lydia Turner and Prof. Richard Eckard.

I would like to acknowledge the support of the Tasmanian Institute of Agriculture (TIA), University of Tasmania, for employment when these publications were written and allocating a small proportion of my work time to prepare this PhD thesis for submission. During the last eight years, the published papers presented in this thesis were externally funded by various organisations. Their support is gratefully appreciated and mentioned in each publication as applicable.

My love of agriculture, and in particular dairying, is in my blood. Thankyou Grandma and Pamp for watching over me and giving me a love of all things bovine.

Thankyou Prof. Danny Donaghy, my TIA supervisor in the mid 2000's who inspired me to complete my Honours and once completed, encouraged me to continue on to become Dr Kaz. Well it may be more than a decade later, but it has finally happened Danny!

To Prof. Richard Eckard and Mrs Cathy Phelps who, over the space of a weekend, came up with a small 8-month project reviewing the GHG emissions of three typical Australia dairy farms. Who would have thought that we would be still working in this field more than a decade later? Thank you for starting the journey allowing me to build expertise, skills and knowledge in dairy GHG emissions.

Thank you to my friends and family who have asked me over the last decade "what do I do for work?" I have never been great at trumpeting my expertise, just ask my wife. I have always enjoyed educating those that taken the time to listen and learn. My favourite question to date relating to GHG emissions was about the ‘farting’ cow. Enlightening people that cows don’t fart, well they do, but most of the methane comes out of their mouths and not their rear end, has been a lot of fun.

A huge thankyou to my parents and siblings for always being there in the good and tough times. I know when I first graduated with my Bachelor’s Degree from the University of Western Sydney- Hawkesbury campus back in the mid 1990’s, my mum was a bit shocked that after three years of tertiary education, I went milking cows. My time at University and then working on commercial dairy farms, here and abroad, was the foundation of my career that has taken me around the world, has given me lifetime friends, some of which have the bonus of being called family. My decades of working within and for the dairy industry has resulted in this PhD.

Finally, and most importantly, I would like to thank my wife Annie. Your encouragement, support and enthusiasm throughout my career, and in particular your patience and understanding over the last 12 to 18 months has been invaluable. We have finally made it my gorgeous Boo.

## Statement pertaining to this thesis

This thesis is written based on published works from research projects conducted by the PhD candidate and is presented in accordance with the requirements set by the University of Tasmania for PhD by Prior Publication. The publishers of the papers comprising Chapters 3 to 7 hold the copyright for that content, and access to the material should be sought from the respective journals. The remaining non-published content of the thesis may be made available for loan and limited copying and communication in accordance with the Copyright Act 1968.

The thesis includes:

- Clear and detailed statements on authorship;
- A comprehensive and concise introduction to the work showing how the individual publications are linked by a common theme;
- A critical review of literature relating to dairy greenhouse gas (GHG) emissions and potential mitigation options, including sections of my published papers;
- The five prior-published peer reviewed publications in full. Although the individual publications arising from this thesis are retained in the format prescribed by the respective journals (*e.g.* American spelling in Chapter 3), the publications are formatted in a consistent uniform style throughout in standard typescript;
- A general discussion chapter;
- A conclusion chapter;
- A single reference list listing all references used throughout the thesis;
- An appendix containing a table indicating the number of citations for the five published papers, list of other peer-reviewed publications, book chapters and conference proceedings from the last eight years not included as research chapters in this thesis, although contributing to climate change adaptation and GHG mitigation of Australian livestock industries;
- An appendix containing background information and results from the MACC analysis.

## Statement of Co-authorship

The following people and institutions contributed to the publication of work undertaken as part of this thesis:

- Ms Karen Christie, Tasmanian Institute of Agriculture, Burnie, University of Tasmania (the candidate);
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- Mrs Catherine Phelps, Dairy Australia, Southbank, Victoria (Author 7).

This thesis contains five publications prior to the completion of the candidature. These are listed below, with authorship, title of publication, journal details and contribution to each publication by the candidate and co-authors.

**Publication 1:** Located in Chapter 3

**Christie KM**, Rawnsley RP, Eckard RJ (2011). A whole farm systems analysis of greenhouse gas emissions of 60 Tasmanian dairy farms. *Animal Feed Science and Technology* **166-167**, 653-662

- Candidate contributed 65%: developed the concept, conducted all the experimental work, analysed the data and wrote most of the manuscript
- Author 1 contributed 20%: assisted with the concept, advised with statistical analysis and revised the manuscript
- Author 2 contributed 15%: assisted with concept and revised the manuscript

**Publication 2:** Located in Chapter 4

**Christie KM**, Gourley CJP, Rawnsley RP, Eckard RJ, Awty IM (2012). Whole-farm systems analysis of Australian dairy farm greenhouse gas emissions. *Animal Production Science* **52**, 998-1011

- Candidate contributed 70%: developed the concept, conducted all the experimental work, analysed the data and wrote most of the manuscript
- Author 1 contributed 10%: assisted with the concept and revised the manuscript
- Author 2 contributed 10%: assisted with the concept and revised the manuscript
- Author 3 contributed 5%: contributed farm data
- Author 4 contributed 5%: contributed farm data

**Publication 3:** Located in Chapter 5

**Christie KM**, Rawnsley RP, Harrison MT, Eckard RJ (2014). Using a modelling approach to evaluate two options for improving animal nitrogen use efficiency and reducing nitrous oxide emissions on dairy farms in southern Australia. *Animal Production Science* **54**, 1960-1970

- Candidate contributed 70%: developed the concept, conducted most of the experimental work, analysed the data and wrote most of the manuscript
- Author 1 contributed 15%: assisted with the concept, statistical analysis and revised the manuscript
- Author 2 contributed 10%: assisted with the concept and revised the manuscript
- Author 5 contributed 5%: revised the manuscript

**Publication 4:** Located in Chapter 6

**Christie KM**, Harrison MT, Trevaskis LM, Rawnsley RP, Eckard RJ (2016). Modelling enteric methane abatement from earlier mating of dairy heifers in subtropical Australia by improving diet quality. *Animal Production Science* **56**, 565-573

- Candidate contributed 70%: developed the concept, conducted most of the experimental work, analysed the data and wrote most of the manuscript
- Author 1 contributed 10%: assisted with the concept, assisted with the experimental work and revised the manuscript
- Author 6 contributed 10%: assisted with the concept and revised the manuscript
- Author 2 contributed 5%: revised the manuscript
- Author 5 contributed 5%: revised the manuscript

**Publication 5:** Located in Chapter 7

**Christie KM**, Rawnsley RP, Phelps C, Eckard RJ (2018). Revised greenhouse-gas emissions from Australian dairy farms following application of updated methodology. *Animal Production Science* **58**, 937-942

- Candidate contributed 75%: developed the concept, conducted all experimental work, analysed the data and wrote the manuscript
- Author 1 contributed 10%: revised the manuscript
- Author 2 contributed 10%: revised the manuscript and advised statistical analysis
- Author 7 contributed 5%: assisted with the concept

We, the undersigned, endorse the above stated contribution of work undertaken for each of the published peer-reviewed manuscripts contributing to this thesis:

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# Table of Contents

## Abbreviations

| | |
|---|---|
| °C | Degrees Celsius |
| 3NOP | 3-Nitrooxypropanol |
| A4N | Accounting4Nutrients |
| ADCC | Australian Dairy Carbon Calculator |
| AFOLU | agriculture, forestry and other land use |
| ANOVA | analysis of variance |
| BoM | Bureau of Meteorology |
| BW | bodyweight |
| C | carbon |
| CCC | Concordance Correlation Coefficient |
| CFI | Carbon Farming Initiative |
| $CH_4$ | methane |
| $CO_2$ | carbon dioxide |
| $CO_2e$ | carbon dioxide equivalents |
| CP | crude protein |
| CSIRO | Commonwealth Scientific and Industrial Research Organisation |
| CT | condensed tannin |
| DAFF | Australian Department of Agriculture, Fisheries and Forestry |
| DCD | dicyandiamide |
| DDMI | digestible dry matter intake |
| DGAS | Dairy Greenhouse gas Abatement Strategies |
| DM | dry matter |
| DMD | dry matter digestibility |
| DMI | dry matter intake |
| DoE | Australian Department of the Environment |
| DoEE | Australian Department of the Environment and Energy |
| ECM | energy-corrected milk |
| EF | emission factor |
| EI | emissions intensity |
| ERF | Emissions Reduction Fund |
| FAO | Food and Agriculture Organization of the United Nations |

| | |
|---|---|
| FCE | feed conversion efficiency |
| FPCM | fat and protein-corrected milk |
| FS | farming system |
| FU | functional unit |
| GHG | greenhouse gas |
| GWP | global warming potential |
| H | hydrogen |
| HRCT | high-rainfall cool temperate |
| HRT | high-rainfall temperate |
| IDF | International Dairy Federation |
| IPCC | Intergovernmental Panel on Climate Change |
| ISO | International Organization for Standardization |
| K | potassium |
| LAC | Latin America |
| LUC | Land Use Change |
| LULUCF | Land Use, Land-Use Change and Forestry |
| LW | liveweight |
| LWG | liveweight gain |
| MACC | Marginal Abatement Cost Curve |
| MCF | methane conversion factor |
| ME | metabolisable energy |
| MEI | metabolisable energy intake |
| MMS | manure management system |
| MLA | Meat & Livestock Australia |
| MN | milk nitrogen |
| MRT | medium-rainfall temperate |
| MS | milksolids |
| N | nitrogen |
| NDF | neutral detergent fibre |
| NENA | Near East and North Africa |
| NGER | National Greenhouse and Energy Reporting |
| NGGI | National Greenhouse Gas Inventory |
| $NH_3$ | ammonia |
| $NH_4$ | ammonium |

| | |
|---|---|
| NI | nitrification inhibitor |
| $NO_3$ | nitrate |
| NORMDIST | normal distribution |
| NUE | nitrogen use efficiency |
| $N_2O$ | nitrous oxide |
| OM | organic matter |
| P | phosphorus |
| S | sulphur |
| SGS | Sustainable Grazing Systems |
| SMLR | stepwise multiple linear regression |
| SN | supplementary feed nitrogen |
| SSA | Sub-Saharan Africa |
| UI | urease inhibitor |
| UNDESA-PD | United Nations Department of Economic and Social Affairs, Population Division |
| UNFCCC | United Nations Framework Convention on Climate Change |
| WCS | whole cottonseed |
| WFPS | water-filled pore space |
| WSC | water-soluble carbohydrate |

Note: Chapter 4 contains a series of abbreviations as part of the equations and are explained as required in that chapter as opposed to listed here.

## Erratum

*Chapter 5*

An error appeared in the peer reviewed published paper in section 5.2.2.4. The text in this thesis has been altered to read "Denitrication losses are from the soil $NO_3$ pool".

# List of Figures

# List of Tables

# CHAPTER 1 INTRODUCTION

## 1.1 Global greenhouse gas emissions

The world is currently facing many challenges, including how we can simultaneously feed an increasing global population while reducing greenhouse gas (GHG) emissions that are contributing to global warming. The current population of 7.6 billion people is predicted to increase to 9.8 billion people by 2050 and possibly up to 13.2 billion people by 2100 (UNDESA-PD, 2017). The three major gases that are widely accepted as contributing to global warming are carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$). Carbon dioxide emissions have risen from 278 ppm at the start of the industrial revolution in the mid-18$^{th}$ century, to 391 ppm in the early 2010's (IPCC, 2013), increasing above 400 ppm, for the first time, in 2016 (Blunden and Arndt, 2017). Emissions of $CH_4$ and $N_2O$, which are more directly linked to agriculture, have also increased by 150 and 20%, respectively, over this same period (IPCC, 2013). Carbon dioxide equivalents ($CO_2e$) are used to compare the emissions of these three gases from various sources, based on their global warming potential (GWP). Methane and $N_2O$ have 28 and 265 times the radiative force of $CO_2$, respectively (IPCC, 2014a).

It is estimated that global GHG emissions in 2016, excluding Land Use, Land-Use Change and Forestry (LULUCF), totalled 49.3 gigatonnes of $CO_2e$ (Olivier *et al.*, 2017). The Intergovernmental Panel on Climate Change (IPCC) was established in 1988 to consider the need for a framework for scientific and environmental assessments of all aspects of the GHG issue (IPCC, 1997). The first assessment report of the IPCC served as the basis for negotiating the United Nations Framework Convention on Climate Change (UNFCCC). One of the first tasks of the UNFCCC was to establish national inventories of GHG emissions and removals. These inventories were used to create the 1990 benchmark and subsequent annual reporting for countries reporting under the Kyoto Protocol. The IPCC was tasked with developing good practice guidelines for national GHG inventories. Under the IPCC guidelines, countries could elect to use the Tier 1 default calculations and emissions factors, or their own country-developed Tier 2 or Tier 3 methodologies, underpinned by local research subject to IPCC review.

## 1.2 Australia's greenhouse gas emissions

Australia's GHG emissions were estimated as 533 Mt $CO_2$e in 2016, after taking into consideration the avoidance/sequestration of 16 Mt $CO_2$e attributed to LULUCF, (DoEE, 2018). The stationary energy sector was the largest source of GHG emissions, accounting for over half of this total. After transport, at 18% of national GHG emissions, agriculture was the third-largest contributor, emitting 69 Mt $CO_2$e and thus accounting for 13% of the nation's GHG emissions (DoEE, 2018). The major source of agricultural emissions was enteric $CH_4$ fermentation from the livestock industries (predominantly dairy cattle, beef cattle and sheep), at 72% of national agricultural emissions (DoEE, 2018). Agricultural soils, dominated by direct and indirect $N_2O$ emissions from organic and inorganic nitrogen (N) fertilisers, animal waste and crop residues, accounted for 19% of national agricultural emissions (DoEE, 2018). Emissions from manure management accounted for 5% of agricultural emissions with minor emissions from lime ($CaCO_3$) and urea ($CH_4N_2O$) applications, field burning of crop residues and rice cultivation (DoEE, 2018).

## 1.3 Australia's dairy industry

The dairy industry is one of Australia's major rural industries, ranked third behind beef and wheat, with a farmgate value of AU $4.3 billion in 2017/18 (Dairy Australia, 2018). While the bulk of milk production occurs along the coastal areas of the south-east corner of the country, the industry is also located in sub-tropical coastal regions of Queensland and northern New South Wales, the south-west of Western Australia, and adjacent to major river systems in inland New South Wales, South Australia and Victoria (Figure 1.1). In 2017-18, the dairy industry produced 9.3 billion litres from 1.56 million cows across 5,700 farms (Dairy Australia, 2018). Victoria alone produced two-thirds of the nation's milk production, with New South Wales and Tasmania producing 12 and 10% of the nation's milk production, respectively (Dairy Australia, 2018).

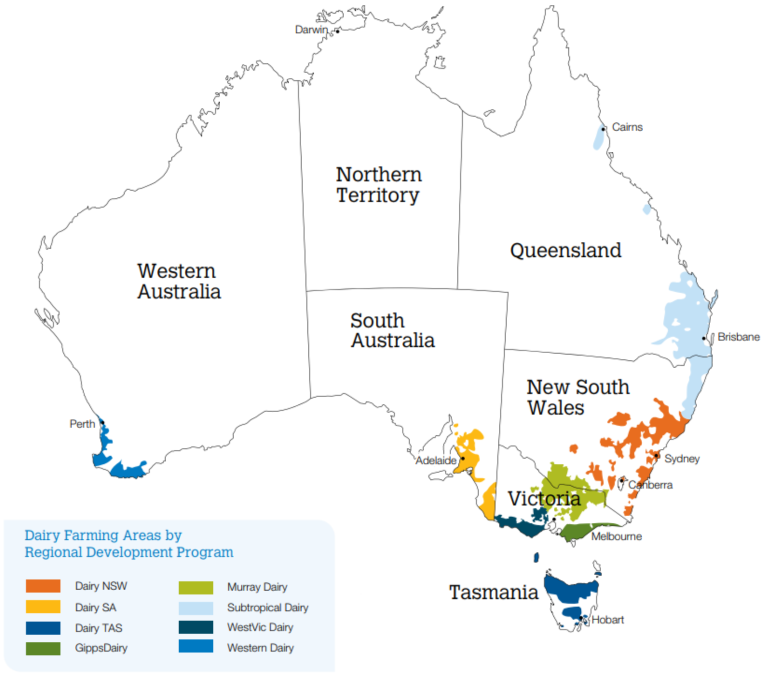


**Figure 1.1** Major dairying regions of Australia separated into the regional development program areas (Dairy Australia, 2018).

The Australian dairy industry can be broadly classified into five farming systems (FS) based on the level of supplementary feeding, the feeding method and infrastructure necessary for supplementary feeding and the number of months of the year in which pasture is grazed (Dairy Australia, 2015a). These FSs are:

- FS1- year-round grazed pasture and other forages fed in the paddock, and up to 1 t dry matter (DM) of grain/concentrate fed in the milking parlour;
- FS2- year-round grazed pasture and other forages fed in the paddock, and > 1 t DM of grain/concentrate fed in the milking parlour;
- FS3- year-round grazed pasture and other forages or partial mixed ration fed on a feed pad, and > 1 t DM of grain/concentrate fed in the milking parlour;
- FS4- pasture grazed for < 9 months per annum with a partial mixed ration fed on the feed pad, and grain/concentrate fed in the milking parlour;
- FS5- zero grazing with cows housed and fed a total mixed ration on a feed pad year-round, and grain/concentrate fed in the milking parlour and/ or the total mixed ration.

Based on the 2015 National Dairy Farmer Survey, most Australian farms were classified as FS1 and FS2 farms, at 23 and 64%, respectively (Dairy Australia, 2015a). However, there were regional differences, with practically all the Tasmanian industry classified as either FS1 or FS2 farms (1% not considered FS1 or FS2), and 27% of the industry in the Subtropical dairy region (encompassing northern NSW and all of Queensland (Figure 1.1)) classified as either FS3, FS4 or FS5 farms (Dairy Australia, 2015a).

Prior to 2011, there was a lack of GHG emissions data for the Australian dairy industry at the individual farm scale. Therefore, the estimation of the emissions intensity (EI) of milk production and any link between individual farm GHG emissions and key farm parameters had not been established. The contribution of individual sources of GHG emissions, as a proportion of total farm emissions, had also not been ascertained for the Australian dairy industry. These are all critical to understand and inform potential mitigation options to reduce on-farm GHG emissions.

From a practical perspective, measuring on-farm GHG emissions is difficult, time consuming, expensive and can only consider a small combination of variables (Bryant *et al.*, 2011; Smith and Western, 2013). Modelling can provide an alternative to field experiments for exploring productivity and environmental impacts over a longer time frame and review potential mitigation options that incorporate the influence of climatic conditions and include more dynamic and complex management practices (Eckard *et al.*, 2014; Jones *et al.*, 2017).

## 1.4 Thesis aims

Using a modelling approach, the following hypotheses were tested in this thesis:

1. The EI of milk production for Tasmanian dairy farms is comparative to other pasture-based systems throughout the developed world;
2. The EI of milk production for Australian farms is similar to those in Tasmania and other pasture-based systems throughout the developed world;
3. That FS influences the EI of milk production for Australian dairy farms;

4. Changes to the Australian National Greenhouse Gas Inventory (NGGI) methodology for estimating dairy farm GHG emissions will result in no difference in the EI of milk production for Australian dairy farms;
5. A greater reduction in $N_2O$ emissions and the EI of milk production can be achieved by better balancing the energy to N ratio in the milking cow's diet than increasing the N captured in milk;
6. Improving the energy density of the diet of heifers will increase their liveweight gain (LWG) between weaning and first mating and thus reduce their cumulative enteric $CH_4$ emissions and EI (kg $CO_2$e/kg liveweight (LW)) over this time period.

The research used to test each of the abovementioned hypotheses is detailed in Chapters 3 to 7.

## 1.5 Thesis structure

Chapter 2 contains a literature review describing global and national GHG emissions, Australian dairy GHG emissions, a comparison of dairy emissions from throughout the world, aspects to consider when comparing dairy emissions and a review of mitigation options to reduce $CH_4$ and $N_2O$ emissions. The review of mitigation strategies is not an exhaustive list, rather an indication of the scope of options available to dairy farmers with results from national and international studies presented. As this is a thesis to attain a PhD by Prior Publication, the literature review does refer to research from Chapters 3 to 7, highlighting gaps that were present in our knowledge at the time of publishing these papers.

Chapters 3 to 7 are prior published peer-reviewed publications that have used GHG accounting and modelling to advance our knowledge with respect to Australia's dairy GHG emissions and highlight two potential mitigation strategies to reduce the EI of milk production for the dairy industry. During the 2000's, analyses of the GHG emissions of dairy farm systems had been undertaken in other countries, including Germany (Haas *et al.*, 2001), Sweden (Cederberg and Flysjö, 2004), Ireland (Casey and Holden, 2005a) and New Zealand (Basset-Mens *et al.*, 2005). However, there was a paucity of whole farm systems studies undertaken for Australian conditions. The development of the Dairy Greenhouse gas Abatement Strategies (DGAS) calculator in the late 2000's based on the Australian NGGI methodology, facilitated the estimation of individual dairy farm GHG emissions (Christie *et al.*, 2008).

Chapter 3 was the first published examination of Australian whole-farm dairy GHG emissions using DGAS, focusing on the emissions of 60 Tasmanian dairy farms (Christie *et al.*, 2011). These farms were all grazed-pasture based (FS1 and FS2 farms), reflective of the industry in Tasmania. A linear relationship between total farm GHG emissions and milk production, milking herd size and total farm area were estimated. The proportion of emissions from each source, such as from enteric $CH_4$ or $N_2O$ from N fertilisers, was determined. In addition, key farm parameters were modelled using a step-wise multiple linear regression (SMLR) analysis to ascertain which parameters could explain milk EI (kg $CO_2$e/kg fat and protein-corrected milk (FPCM)), cow EI (t $CO_2$e/milker) and farm area EI (t $CO_2$e/ha).

Given the diversity of the industry throughout Australia, both from geographic and management perspectives, Chapter 4 expanded the study region to examine the GHG emissions of 41 Australian dairy farms, representing all states of Australia, using the same methodology as Chapter 3 (Christie *et al.*, 2012). Farm location introduced a diversity of feeding options not present in the Tasmanian industry. Examples include:

- greater reliance on supplementary feed being delivered as a partial mixed ration on a feedpad;
- a higher proportion of the herd's grain requirements produced on-farm (*e.g.* some Queensland farms);
- farms closer to populous areas having access to waste products from other food manufacturing industries to form part of the herd's diet (*e.g.* some Victorian and South Australian farms feeding distiller's grain).

Chapter 4 explored all the analyses undertaken when examining GHG emissions of Tasmanian dairy farms in Chapter 3. In addition, Chapter 4 introduced the concept of FS and explored relationships between FS and EI of milk production.

In the 2000's there were many reviews examining mitigation options to reduce livestock GHG emissions (Dalal *et al.*, 2003; Kebreab *et al.*, 2006; Beauchemin *et* *al.*, 2008; Eugéne *et al.*, 2008). However, there were few mitigation studies examining component GHG emissions options for the Australian dairy industry. Chapters 5 and 6 modelled targeted GHG mitigation strategies for the dairy industry.

Chapter 5 evaluated two options for improving animal nitrogen use efficiency (NUE) and reducing $N_2O$ emissions for the milking cow across three dairying regions of south-eastern Australia (Christie *et al.*, 2014). These options were altering the supplementary feed N (SN) concentration of the diet, as a surrogate of overall dietary N concentration, and altering the N concentration in milk (MN), as a surrogate for breeding animals with greater protein concentration in their milk.

Chapter 6 examined the effect of earlier mating of dairy heifers in subtropical Australia by improving diet quality and its impact on reducing enteric $CH_4$ from the period between weaning and first mating (Christie *et al.*, 2016). Two forms of analysis were undertaken. In the static approach, feed nutritional characteristics were constant over the duration of the study. In the dynamic approach, using the Sustainable Grazing Systems (SGS) model (Johnson, 2013), feed nutritional characteristics varied depending on seasonal climatic conditions and pasture availability.

In 2015, the NGGI methodology underwent a major review, with changes to algorithms and emission factors (EFs) altered to reflect recent Australian-specific research. In addition, the review signified changes to animal and feed characteristics that had occurred since the original methodology was developed in the early 1990's and ensured the Australian inventory remained reflective of IPCC and UNFCCC changes. These changes were incorporated into the DGAS calculator, with the calculator rebranded the Australian Dairy Carbon Calculator (ADCC). Chapter 7 re-visited the 41 Australian dairy farms previously examined in Chapter 4 to review the effect of the updated NGGI changes on EI of milk production for each dairying state of Australia and the proportion of emissions from each source (Christie *et al.*, 2018). A Concordance Correlation Coefficient (CCC) examined the agreement between the old and new NGGI methodologies with respect to individual farm EI, regional EI and the proportion of emissions from each source.

The focus of the research chapters were assessments of the GHG emissions and potential mitigation options for Australian dairy farms. In many ways, the focus was on the EI of milk production; a valid temporal and spatial comparison. However, what the globe is experiencing is a net increase in GHG emissions. This disconnect between EI and net emissions needs to be addressed if we are to restrict the increase in mean global temperature to < 2°C as agreed to at the UNFCCC's Conference of the Paris Agreement (COP21) meeting in December 2015 (European Commission, 2015). The discussion chapter contains a Marginal Abatement Cost Curve (MACC) analysis to evaluate the GHG emissions reduction and the farm productivity and profitability changes associated with implementing seven currently available mitigation options across four contrasting Australian dairy farms. A comparison of each of the seven mitigation options was undertaken for four contrasting case study dairy farms so that the combination of FS with EI reflected the variation in EIs across the individual farm results from Chapters 4 and 7.

In addition, the discussion chapter addresses the need to fully explore all implications of mitigation options, such as the ethics of feeding grain to livestock to assist in reducing emissions versus grain for human consumption, the importance of a global carbon (C) market in addressing aspects such as leakage, the role of the consumer in demanding products with lower emissions and who is ultimately going to pay for abatement of emissions.

## 1.6 General conclusions

In conclusion, the research undertaken in this body of published work has improved our understanding of the GHG emissions profile of the Australian dairy industry. The development and use of the ADCC (and its predecessor DGAS) has given researchers, farm advisers and farmers an estimation of individual farm GHG emissions. Total farm annual milk production has been identified as a potential surrogate to estimate total farm GHG emissions, although the EI of milk production, on an individual farm basis at the national scale, varied by over 100%. In addition, the research has given the industry an improved understanding of the key drivers of EI of milk production and highlighting a link between EI and FS. Two specific mitigation options were explored in detail in this thesis. This work highlighting the importance of improving dietary N balance, offered to the milking cow, to reduce $N_2O$ emissions, while improving the diet quality of growing heifers was shown to significantly reduce enteric $CH_4$ emissions between weaning and first mating. Given that these two mitigation options target different aspects of the whole-farm system, they are likely to be additive, with little to zero downside risk. Modified versions of these two mitigation options, combined with five other currently available mitigation options, were then explored using a MACC analysis to evaluate the GHG emissions reduction and cost-benefit across four representative Australian dairy farms. All mitigation options examined reduced total farm GHG emission, with three of the seven options also profitable across all four farms. The modelling undertaken in the thesis has also provided a knowledge base from which other win:win mitigation options can be explored and promoted.

# CHAPTER 2 LITERATURE REVIEW

## 2.1 Global greenhouse gas emissions

"Anthropogenic greenhouse gas emissions have increased since the pre-industrial era, driven largely by economic and population growth, and are now higher than ever. This has led to atmospheric concentrations of carbon dioxide, methane and nitrous oxide that are unprecedented in at least the last 800,000 years. Their effects, together with other anthropogenic drivers, have been detected throughout the climate system and are *extremely likely* to have been the dominant cause of the observed warming since the mid-20$^{th}$ century" (IPCC, 2014a).

The three major gases that are widely accepted as contributing to global warming are $CO_2$, $CH_4$ and $N_2O$. Since the 1850's, global $CO_2$ has increased from 278 to 391 ppm, increasing above 400 ppm, for the first time, in 2016 (Blunden and Arndt, 2017). Methane has increased from around 800 to 1,800 ppb and $N_2O$ has increased from around 270 to 320 ppb (IPCC 2014a; Figure 2.1).

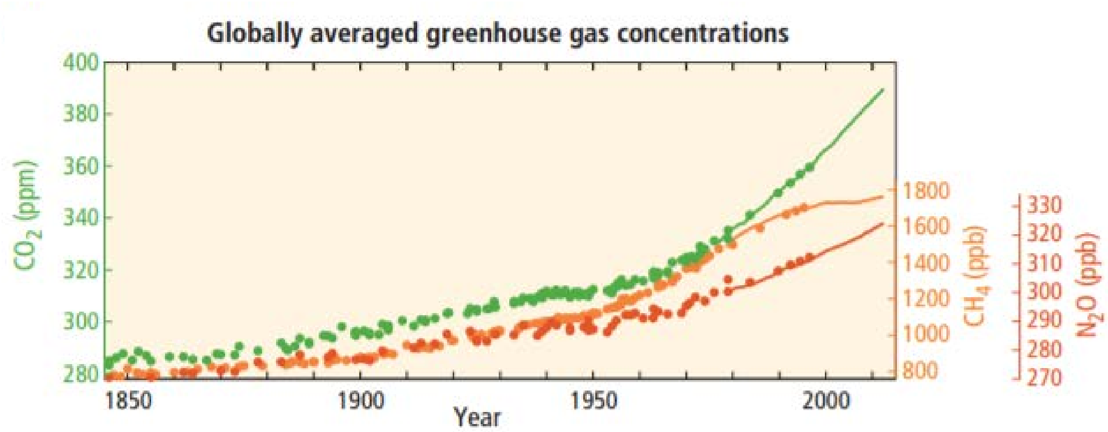


**Figure 2.1** Atmospheric concentrations of the greenhouse gases carbon dioxide ($CO_2$, green), methane ($CH_4$, orange) and nitrous oxide ($N_2O$, red) determined from ice core data (dots) and from direct atmospheric measurements (lines) (IPCC, 2014a).

Carbon dioxide equivalents are used to compare the emissions of these three gases from various sources, based on their global warming potential (GWP). Methane and $N_2O$ have 28 and 265 times the radiative force of $CO_2$, respectively (IPCC, 2014a). It is estimated that in 2016, global GHG emissions, excluding LULUCF, totalled 49.3 Gt $CO_2$e (Olivier *et al.*, 2017). Based on 2010 figures, $CO_2$e from electricity and heat production was the biggest contributor to global GHG emissions, at 25% (IPCC, 2014b). The next largest contributor was agriculture, forestry and other land use (AFOLU sector) at 24% of global GHG emissions, with most of this attributed to agriculture (IPCC, 2014b). Industrial manufacturing (*e.g.* production of chemicals, iron, steel, cement, plastics etc) was the third largest contributor, at 21%, followed by transportation at 14%, other energy at 10% and buildings at 6% (Figure 2.2; IPCC, 2014b).

All GHG emissions are considered either Scope 1 (emissions released to the environment as a direct result of activity, *e.g.* agriculture), Scope 2 (emissions released to the environment from the generation and consumption of energy) or Scope 3 (indirect emissions not included in Scope 2 that are generated in the wider economy) (NGER, 2018). Emissions allocated to electricity and heat production are indirect $CO_2$ emissions (Scope 2) and as such could be allocated to their primary source (Scope 1). For example, electricity and heating used within the dairy industry to milk cows and irrigate are considered Scope 2 emissions. The $CO_2$ emitted with the consumption of electricity is not attributed to the agricultural sector, it is reported within the electricity and heat production sector. For agriculture, the indirect $CO_2$ emissions (Scope 2 and 3 emissions) is minimal at <1% of global emissions (Figure 2.2; IPCC, 2014b).

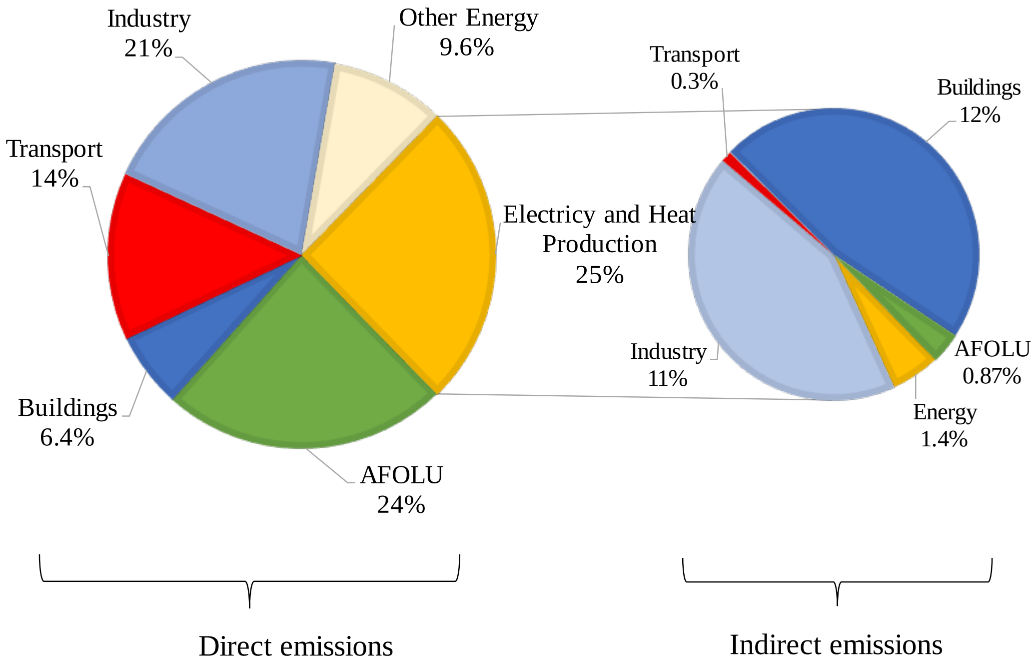


**Figure 2.2** Global greenhouse gas emissions by economic sector in 2010, with agriculture included in the AFOLU (agriculture, forestry and other land use) sector. The left circle shows direct greenhouse gas emission shares of the five economic sectors. The right circle shows how indirect carbon dioxide ($CO_2$) emissions from electricity and heat production are attributed to the sectors of final energy use (adapted from IPCC, 2014b).

The IPCC was established in 1988 to consider the need for a framework for scientific and environmental assessments of all aspects of the GHG issue (IPCC, 1997). The first assessment report of the IPCC served as the basis for negotiating the UNFCCC. One of the first tasks of the UNFCCC was to establish national inventories of GHG emissions and removals. These inventories were used to create the 1990 benchmark and subsequent annual reporting for countries reporting under the Kyoto Protocol. The IPCC were tasked with developing good practice guidelines for national GHG inventories. Under the IPCC guidelines, countries could elect to use the Tier 1 default calculations and EFs, or their own country-developed local Tier 2 or Tier 3 methodologies and EFs, underpinned by local research subject to IPCC review.

Unquestionably, the livestock sector represents a significant source of GHG emissions worldwide, by emitting GHG emissions either directly (*e.g.* enteric fermentation and manure management) or indirectly (*e.g.* from feed-production activities and the conversion of forest into pasture). Based on a life cycle assessment, the contribution of livestock to global anthropogenic GHG emissions represents 14.5% (Gerber *et al.*, 2013a). The IPCC follows a different attribution procedure, reporting agriculture, including non-livestock emissions such as those from crop production, as contributing to 12% of global anthropogenic emissions; increasing to a startling 30% when land use and land use change (LUC) is included (Smith *et al.*, 2007). These two examples highlight the difficulty of estimating global GHG emissions, but both underline the contribution of agriculture to the global issue.

## 2.2 National greenhouse gas emissions

In meetings its reporting obligations under the UNFCCC, annually since 1990, the Australian Federal Government has estimated the nation's GHG emissions. This is undertaken using Tier 1 and country-specific Tier 2 methodologies and EFs. The NGGI methodologies evolve over time and continue to be refined as new scientific information emerges and international practices advance. A NGGI methodology update occurred in 2015, reporting emissions for 2013 and retrospectively estimating annual GHG emissions back to 1990.

In 2016, total GHG emissions was estimated at 533 Mt $CO_2$e when taking into consideration avoidance and/or sequestering of C attributed to LULUCF (3%; DoEE, 2018). This represents a decline from 583 Mt $CO_2$e in 1990. Much of this decline was attributed to changes in LULUCF, with C emissions of 163 Mt $CO_2$e/annum in 1990 (*i.e.* loss of C from deforestation, soil C stock change and biomass burning) compared to avoidance and/or sequestering of 16 Mt $CO_2$e/annum in 2016 (*i.e.* accumulation of C with forest management, afforestation and reforestation with aspects such as increased soil C stock change) (DoEE, 2018). In 2016, stationary energy was the largest source, at 54% of Australia's net GHG emissions (DoEE, 2018; Figure 2.3). Transport was second highest, at 18%, followed by agriculture, at 13%, fugitive emissions from fuel at 9%, emissions from industrial processes and product use at 6%, and waste at 2% (DoEE, 2018; Figure 2.3).

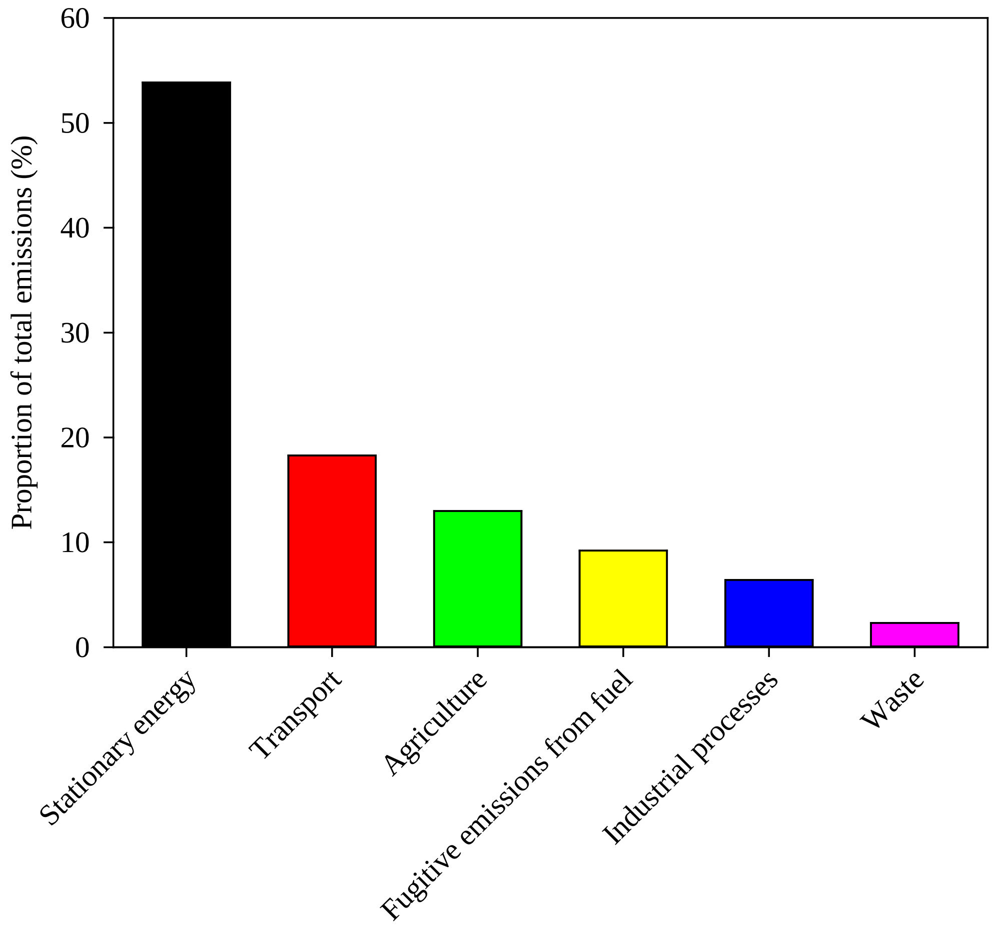


**Figure 2.3** Australia's net greenhouse gas emissions by sector in 2016 (DoEE, 2018).

## 2.3 Australian dairy greenhouse gas emissions

In 2016, Australian agriculture emitted an estimated 69 Mt $CO_2e$ (13% of national GHG emissions; DoEE, 2018). Allocating these emissions to the various agricultural industries is a challenging process and involves some assumptions. For example, enteric $CH_4$ can be easily and clearly allocated to the sector in which it originates from, be it dairy, beef cattle or sheep. In contrast, direct $N_2O$ emissions from inorganic N fertiliser usage is not broken down into specific livestock sectors, but rather the type of system the N fertiliser is applied to, namely irrigated pasture, irrigated crops, non-irrigated pasture, non-irrigated crop, sugar cane, cotton and horticulture/vegetables. Thus, N fertiliser applied to non-irrigated pasture from dairy, beef cattle and sheep are summed together.

Given the high usage of N fertiliser within the dairy industry relative to other livestock industries (Eckard, 2001; Dalal *et al.*, 2003; Gourley *et al.*, 2012b), the assumption made in Figure 2.4 was that the dairy industry was responsible for 95% of the N fertiliser emissions associated with pasture, with the balance allocated to the beef and sheep industries. The N fertiliser applied to irrigated and non-irrigated crops were attributed to cereals such as wheat (*Triticum aestivum*), barley (*Hordeum vulgare*), rice (*Oryza sativa*) and maize (*Zea mays*) (Angus and Grace, 2017), although there is some N fertiliser applied to rainfed and irrigated crops on dairy farms (*e.g.* growing maize for silage conservation). In addition, it was assumed that the dairy industry was responsible for 10% of $CO_2e$ from lime and urea applications, the balance to other livestock, cropping and horticultural industries. Based on these assumptions, the dairy industry was responsible for an estimated 10.0 Mt $CO_2e$, equivalent to 2% of the nation's total emissions and 14% of agricultural emissions (DoEE, 2018; Eckard and Clark, 2018).

Dairy GHG emissions can be broadly grouped into four sources; enteric $CH_4$, waste $CH_4$, waste $N_2O$ and N fertiliser $N_2O$ (Figure 2.4; DoEE, 2018). Some of these can be further broken down into sub-sources (*e.g.* direct and indirect $N_2O$ losses from N fertilisers (Figure 2.4; DoEE, 2018)). In addition, there is a proportion of indirect GHG emissions associated with electricity and fuel consumption (Scope 2 emissions) and the $CO_2e$ emissions associated with the production/manufacturing of key farm inputs such as grains/concentrates, fodder and fertilisers (Scope 1 emissions). While these two indirect sources of GHG emissions are a consequence of the business of dairying, similar to the international reporting mentioned above in Figure 2.2, on the national scale they are accounted for in either another sector, as the case for electricity accounted for in the stationary energy sector, or within another sub-section of the agricultural sector (*i.e.* emissions from growing grain attributed to the cropping farm, not the dairy farm that consumes the grain).

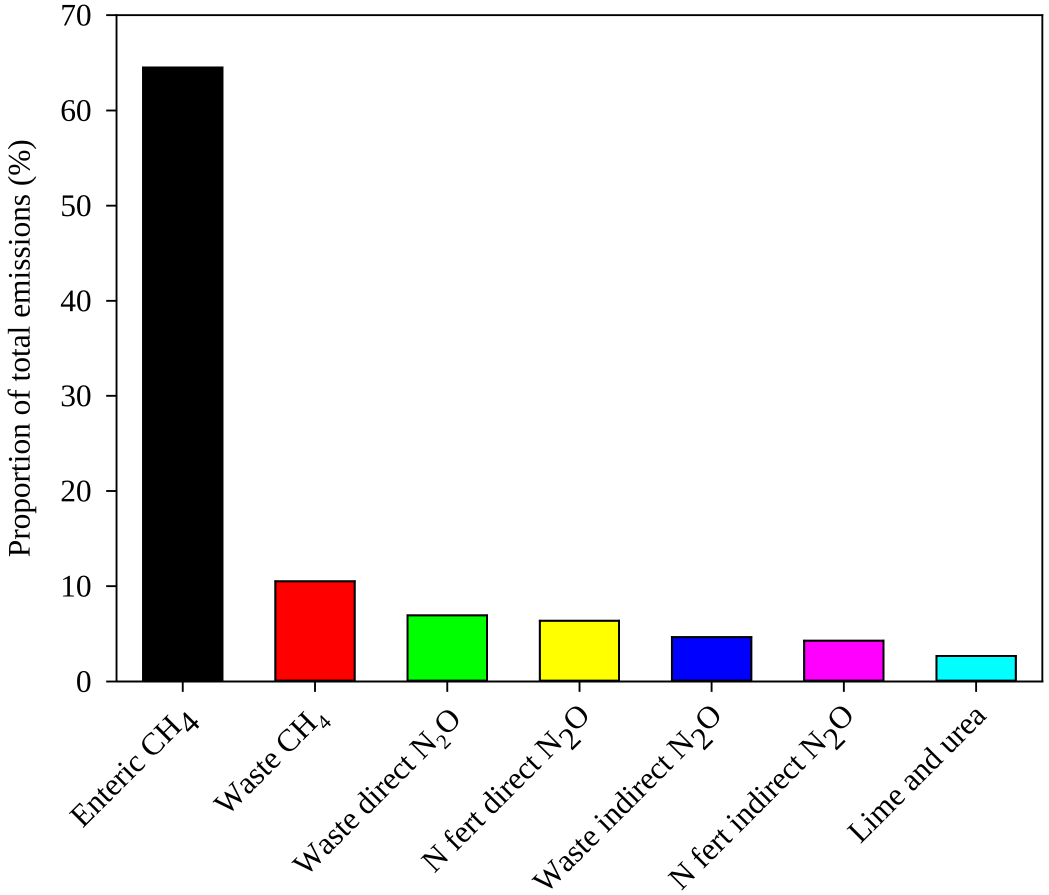


**Figure 2.4** Proportion of on-farm dairy greenhouse gas emissions attributed to each source in 2015, based on the Australian national inventory (DoEE, 2018).

### *2.3.1. Enteric methane emissions*

Nationally, enteric $CH_4$ is the largest source of GHG emissions and accounted for two-thirds of dairy GHG emissions (Figure 2.4; DoEE, 2018). The formation of enteric $CH_4$, known as methanogenesis, mainly occurs in the rumen, and to a lesser extent large intestine, due to the fermentation process (Clark *et al.*, 2005; McAllister and Newbold, 2008). Cellulose is broken down into volatile fatty acids through microbial activity, which releases hydrogen (H) ions (Boadi *et al.*, 2004; Iqbal *et al.*, 2008). Methanogenesis occurs when the single-celled micro-organisms called Archaea utilises the $H_2$ with $CO_2$ to form $CH_4$ under anaerobic conditions (Boadi *et al.*, 2004; Attwood *et al.*, 2011). Methanogenesis results in the loss of 6-10% of gross energy intake (Cottle *et al.*, 2011; Hristov *et al.*, 2013a) and is produced at a rate of between 14 and 26 g $CH_4$/kg DM intake (DMI; Kirchgessner *et al.*, 1991; Hristov *et* *al.*, 2013a). Within the Australian NGGI methodology, enteric $CH_4$ emissions is estimated as 20.7 g $CH_4$/kg DMI (Charmley *et al.*, 2016).

### *2.3.2. Waste methane emissions*

Methane emissions from waste (dung and urine deposition) management accounted for 11% of national dairy GHG emissions (Figure 2.4; DoEE, 2018). Most of the $CH_4$ emissions from waste is produced under anaerobic conditions during storage. Within the Australian NGGI methodology, waste $CH_4$ emissions is a function of volatile solids production, several constants and an integrated $CH_4$ conversion factor (MCF) weighted based on the proportion of waste allotted to various manure management systems (MMS) and their corresponding MCF (Christie *et al.*, 2012; DoEE, 2018).

### *2.3.3. Waste nitrous oxide emissions*

Nitrous oxide emissions from waste (dung and urine deposition) management accounted for 12% of national dairy GHG emissions (Figure 2.4; DoEE, 2018). The N in animal waste is either in organic form (dung) or inorganic form (urine). Emissions of $N_2O$ are largely a result of two soil microbial processes, nitrification and denitrification. Nitrification is an aerobic process that oxides ammonium ($NH_4^+$) to nitrate ($NO_3^-$) with denitrification of $N_2O$ a by-product of the process. Denitrification is an anaerobic process that reduces $NO_3^-$ into $N_2$, with $N_2O$ an obligatory intermediate (de Klein and Eckard, 2008). Factors that significantly affect the production of $N_2O$ from animal waste are temperature, water-filled pore space (WFPS), level of organic C, soil pH and soil $NO_3$ (Whitehead, 1995), with soil $NO_3$ levels and soil aeration identified as the most likely key factors affecting $N_2O$ emissions from grazing systems (Eckard *et al.*, 2010). In addition to direct losses of $N_2O$ as described above, a proportion of N lost to the environment through leaching and/or runoff of $NO_3$ and volatilisation of ammonia ($NH_3$) undergoes the same processes when redeposited onto land, resulting in indirect $N_2O$ emissions.

### 2.3.4. *Nitrogen fertiliser nitrous oxide emissions*

Nitrous oxide emissions from N fertiliser management accounted for 10% of national dairy GHG emissions (Figure 2.4; DoEE, 2018). This may be an overestimation given the challenging process of allocating N fertiliser emissions to dairy highlighted above, especially given that the mean $N_2O$ emissions from N fertiliser, reported in Chapter 4, was lower at 3.9%. The transformation and loss processes are the same for N fertiliser as for animal waste. In addition, urea fertiliser and the soil ameliorant lime release $CO_2$ to the atmosphere after application to soil, with emissions from these two totalling 3% of national dairy GHG emissions (Figure 2.4; DoEE, 2018).

## 2.4 Individual farm greenhouse gas emissions

### 2.4.1. *Australian greenhouse gas emission studies*

Prior to 2011, there were no published whole farm systems analyses of Australian dairy farm GHG emissions. However, in 2011, two whole-farm systems studies were published in the same journal issue, Browne *et al.* (2011) and Christie *et al.* (2011). Browne *et al.* (2011) used a whole-farm mechanistic biophysical model to examine the emissions of four representative dairy farms based on regional data for south-western Victoria. The farm was stocked at 1.7 and 2.0 cows/ha fed grazing pastures and consumed either a low (*i.e.* 2-4 kg DM/cow.day) or medium (*i.e.* 4-8 kg DM/cow.day) rate of concentrate supplement. Thirty years of data was collated into an excel spreadsheet, where the output data from the biophysical model (e.g. milk production per cow, feed intake) were incorporated with the national algorithms and EFs to estimate on-farm $CH_4$ and $N_2O$ emissions. The EI of milk production varied between 0.64 and 0.71 kg $CO_2$e/kg FPCM (standardised to 4.0% fat and 3.3% protein; Table 2.1).

Christie *et al.* (2011; Chapter 3) examined the GHG emissions of 60 Tasmanian dairy farms, using the same national algorithms and EFs used by Brown *et al.* (2011) to estimate on-farm $CH_4$ and $N_2O$ emissions estimation. In addition, $CO_2$ emissions from the consumption of energy (electricity and diesel fuel) and pre-farm gate embedded $CO_2$e emissions associated with the production and manufacturing of key farm inputs (*i.e.* grains/concentrates, fodder and fertilisers) were also included in the estimations. This was the first publication examining the GHG emissions of 60 Tasmanian dairy farm businesses, with broad ranges of grain feeding (0 to 2.9 t

DM/cow.lactation), milk production (1,900 to 5,000 litres/cow.lactation), N fertiliser inputs (3 to 414 kg N/ha.annum) and irrigation of the milking platform (0 to 100%; Tables 3.1 to 3.3).

The mean EI of milk production was estimated at 1.04 kg $CO_2$e/kg FPCM and varied between 0.83 and 1.39 kg $CO_2$e/kg FPCM (Table 2.1; Christie *et al.* 2011). This result was greater than the study by Browne *et al.* (2011). However, the Browne *et al.* (2011) study did not include an estimate of emissions from energy consumption and pre-farm gate emissions associated with the production and manufacturing of supplementary feed and fertilisers. These were estimated to contribute an average of an additional 23% in the Christie *et al.* (2011) study. If these sources of emissions were incorporated in the Browne *et al.* (2011) study, the EI of milk production would have increased to between 0.79 and 0.87 kg $CO_2$e/kg FPCM, a result comparative to the lower-end emissions reported by Christie *et al.* (2011).

While there was diversity of farms with the Tasmanian study, all farms were pasture grazing-based systems. Christie *et al.* (2012; Chapter 4) examined the EI of 41 dairy farms, from all regions of Australia, and introduced FS classification into the analysis. Farms varied from predominantly pasture-based through to partial mixed ration farms where the milking cow would spend a proportion of their time on 'hard' surfaces, receiving supplementary feed. Across the 41 Australian dairy farm businesses, there was broad ranges of grain feeding (0 to 2.9 t DM/cow.lactation), milk production (3,250 to 9,870 litres/cow.lactation), N fertiliser inputs (0 to 316 kg N/ha.annum) and irrigation of the milking platform (0 to 82%; Tables 4.1 to 4.3). Christie *et al.* (2012) found that the EI of milk production averaged 1.04 kg $CO_2$e/kg FPCM, ranging between 0.76 and 1.68 kg $CO_2$e/kg FPCM (Table 2.1). This result was comparative to the result of the Tasmanian study, although greater variability between farms was identified.

Gollnow *et al.* (2014) examined 139 Australian dairy farms, using the same Australian NGGI methodology, but with some variation in the EFs for pre-farm gate embedded emissions and the allocation of emissions to milk and meat. The results of their study found the EI of milk production averaged 1.1 kg $CO_2$e/kg FPCM, with 80% of farms varying between 0.9 and 1.4 kg $CO_2$e/kg FPCM range (Table 2.1). While the methodology of estimating total farm GHG emissions was congruent with the above-mentioned studies, there were subtle differences that need to be taken into consideration when comparing results, such as the GWP of the GHG emissions and the inclusion of emissions associated with LULUC with supplementary feeds.

### *2.4.2. International greenhouse gas emission studies*

Results from the aforementioned Australian studies were comparative to results from international studies from developed nations where cows graze pasture either year-round, such as in New Zealand, or for the majority of the year such as in Ireland (Table 2.1). Similarly, results from the Australian studies were comparable to international studies where cattle are housed for most of the year, if not year-round and fed a total mixed ration (*e.g.* USA, Canada, Europe) (Table 2.2). What is clear from the Australian and overseas studies, is that the EI of milk production varies between studies for numerous reasons, some of which are highlighted in Tables 2.1 and 2.2 and explored further here.

**Table 2.1** Dairy greenhouse gas emissions from countries where cattle graze pasture for the majority/all of the year.

| Reference | No. of farms examined | Region/ Country | Sources of emissions examined | Emissions intensity (mean; kg $CO_2$e) | Emissions intensity (range; kg $CO_2$e) | Functional unit | GWP ($CH_4$ & $N_2O$) | Allocation method for milk & meat |
|---|---|---|---|---|---|---|---|---|
| Christie *et al.* (2011) | 60 farms | Tasmania/ Australia | $CH_4$, $N_2O$, $CO_2$ | 1.04 | 0.83 – 1.39 | kg FPCM | 21 & 310 | Milk only |
| Browne *et al.* (2011) | 4 farms | Victoria/ Australia | $CH_4$, $N_2O$ | n/a | 0.64 – 0.71 | kg FPCM | 21 & 310 | Milk only |
| Christie *et al.* (2012) | 41 farms | All regions of Australia | $CH_4$, $N_2O$, $CO_2$ | 1.04 | 0.76 – 1.68 | kg FPCM | 21 & 310 | Milk only |
| Gollnow *et al.* (2014) | 139 farms | Six regions of Australia | $CH_4$, $N_2O$, $CO_2$ | 1.11 | 0.90 – 1.39[a] | kg FPCM | 25 & 298 | Mass |
| Christie *et al.* (2018) | 41 farms | All regions Australia | $CH_4$, $N_2O$, $CO_2$ | 1.07 | 0.84 – 1.54 | kg FPCM | 25 & 298 | Milk only |
| Casey and Holden (2005b) | Average dairy unit | Ireland | $CH_4$, $N_2O$, $CO_2$ | 1.50; 1.30; 1.45 | n/a | kg ECM | 21 & 310 | Milk only; Economic; Mass |
| O'Brien *et al.* (2014a) | 124 farms | Ireland | $CH_4$, $N_2O$, $CO_2$ | 1.11 | 0.87 – 1.72 | kg FPCM | 25 & 298 | Economic |
| Chobtang *et al.* (2016) | 53 farms | New Zealand | $CH_4$, $N_2O$, $CO_2$ | 0.80 | 0.78 – 0.82[b] | kg FPCM | 25 & 298 | Biological |
| de Léis *et al.* (2015) | Typical farm | Brazil | $CH_4$, $N_2O$, $CO_2$ | n/a | 0.74 & 1.01[c] | kg ECM | 25 & 298 | Milk only |
| Galloway *et al.* (2018) | 80 farms | South Africa | $CH_4$, $N_2O$, $CO_2$ | 1.39 | 0.94 – 2.07 | kg FPCM | 25 & 298 | Economic |

[a] 80% confidence interval; [b] 95% confidence interval; [c] Excluding and including land use change. $CH_4$, methane; $CO_2$, carbon dioxide; $CO_2$e, carbon dioxide equivalents; ECM, energy-corrected milk; FPCM, fat and protein-corrected milk; GWP, global warming potential; $N_2O$, nitrous oxide

**Table 2.2** Dairy greenhouse gas emissions from countries where cattle spend the majority of their time confined.

| Reference | No. of farms examined | Country | Sources of emissions examined | Emissions intensity (mean; kg $CO_2e$) | Emissions intensity (range; kg $CO_2e$) | Functional unit | GWP ($CH_4$ & $N_2O$) | Allocation method for milk & meat |
|---|---|---|---|---|---|---|---|---|
| Castanheira *et al.* (2010) | Average dairy farm | Portugal | $CH_4$, $N_2O$, $CO_2$ | 1.02 | n/a | kg raw milk | 25 & 298 | Economic |
| Zehetmeier *et al.* (2014a) | 53 farms | Germany | $CH_4$, $N_2O$, $CO_2$ | 1.02 | 0.79 – 1.25 | kg FPCM | 25 & 298 | Milk only |
| Thomassen *et al.* (2008) | 10 conventional & 11 organic farms | The Netherlands | $CH_4$, $N_2O$, $CO_2$ | 1.4 & 1.5 | 0.1 & 0.3[a] | kg FPCM | 21 & 310 | Mass |
| Nguyen *et al.* (2013) | 6 TMR farms | France | $CH_4$, $N_2O$, $CO_2$ | n/a | 0.87 – 1.62 | kg FPCM | 25 & 298 | Range of methods[b] |
| Kristensen *et al.* (2015) | Typical farm | Denmark | $CH_4$, $N_2O$, $CO_2$ | n/a | 1.20 & 0.81 | kg ECM | 25 & 298 | Nil & biological |
| O'Brien *et al.* (2014b) | TMR farm | UK | $CH_4$, $N_2O$, $CO_2$ | n/a | 0.61 – 0.88 | kg ECM | 25 & 298 | Range of methods[b] |
| Thoma *et al.* (2013) | 536 farms | USA | $CH_4$, $N_2O$, $CO_2$ | 1.23 | 0.7 – 2.7[c] | kg FPCM | 25 & 298 | Biological |
| Vergé *et al.* (2007) | Typical of 5 regions | Canada | $CH_4$, $N_2O$, $CO_2$ | n/a | 0.97 – 1.13 | kg raw milk | 21 & 310 | Milk only |
| de Léis *et al.* (2015) | Typical farm | Brazil | $CH_4$, $N_2O$, $CO_2$ | n/a | 0.54 & 0.78[d] | kg ECM | 25 & 298 | Milk only |

[a] Range of results not presented, however, a standard deviation was presented; [b] Variety of allocation methods including nil, mass, economic, protein, biological, emission or systems expansion; [c] Excluding two outliers beyond this range; [d] Excluding and including land use change. $CH_4$, methane; $CO_2$, carbon dioxide; $CO_2e$, carbon dioxide equivalents; ECM, energy-corrected milk; FPCM, fat and protein-corrected milk; GWP, global warming potential; $N_2O$, nitrous oxide; TMR, total mixed ration

### *2.4.3. Methodology differences*

The single biggest consideration when comparing results between studies is the methodology used to estimate total farm GHG emissions. For example, the GWP of $CH_4$ has increased over time from 21 to 25, thus automatically increasing $CH_4$ emissions by 19% (DoE, 2015). Christie *et al.* (2018; Chapter 7), used the same dataset as from a previous study (Christie *et al.*, 2012; Chapter 4), to examine the effect of changes in Australia's national methodology on GHG emissions. As an average across the 41 dairy farms, the EI of milk production increased by 3%. Using a CCC analysis to assess the agreement between the old and new methodologies, combined with the descriptive scale for the degree of agreement (McBride, 2005), there was moderate agreement (> 0.9) between methodologies for estimating individual-farm EI and almost perfect agreement (> 0.99) for estimating the proportion of emissions from each source. However, there was poor agreement (< 0.9) between methodologies for estimating regional EI, due to changes in manure management EFs between the two methodologies.

Another consideration when comparing results is the level of methodology complexity. The complexity increases from the simplest IPCC Tier 1 level (*e.g.* enteric $CH_4$ at 6.0 ± 0.5% of gross energy intake (IPCC, 2000a), progressing through the country-specific Tier 2 methodology (*e.g.* enteric $CH_4$ for Australian dairy cattle at 20.7 g $CH_4$/kg DMI; Charmley *et al.*, 2016). The most complex is the Tier 3 methodology where the development of sophisticated models consider diet composition in detail, seasonal variation in animal population or feed quality and availability, and possible mitigation strategies (IPCC, 2000a). For example, a study in Canada compared Tier 1 and Tier 2 estimates for enteric $CH_4$ from dairy cattle, finding that the Tier 2 methodology estimate was 6.3% higher than the Tier 1 estimate (Ominski *et al.*, 2007). Most published studies use country-specific Tier 2 EFs, adding further variation in estimations. For example, the Australian emission factor (EF) for the amount of N lost through volatilisation, and converted into $N_2O$, is 0.004 kg $N_2O$-N/kg N (DoE, 2015), compared to the IPCC default factor of 0.01 kg $N_2O$-N/kg N implemented in many other countries, including Canada and New Zealand (Environment and Climate Change Canada, 2017; New Zealand Ministry for the Environment, 2017).

### 2.4.4. System boundary differences

Consistency of system boundary can make comparison of results between studies difficult. For example, in a study by Gerber *et al.* (2013a), the EI of milk production was estimated as 1.6 kg $CO_2$e/kg FPCM for many of the regions of the developed world, a figure greater than many of the studies reported in Tables 2.1 and 2.2. The Gerber *et al.* (2013a) study included emissions from LUC associated with the production of soybean meal (*Glycine max*) as well as emissions associated with post-farm gate (processing of milk into product), potentially contributing an additional 0.2 kg $CO_2$e/kg FPCM, depending on the region (Figure 2.5; Gerber *et al.*, 2013a). Greenhouse gas emissions attributed to LUC were also incorporated in the Australian study by Gollnow *et al.* (2014). For any farm feeding purchased concentrates containing soybean meal or palm kernel extract (*Elaeis guineensis*), the assumption was that these products were imported from South America and SE Asia, respectively. Given that land clearing from forests to agricultural land to grow these crops could not be ruled out, a proportion of GHG emissions were attributed to LUC.

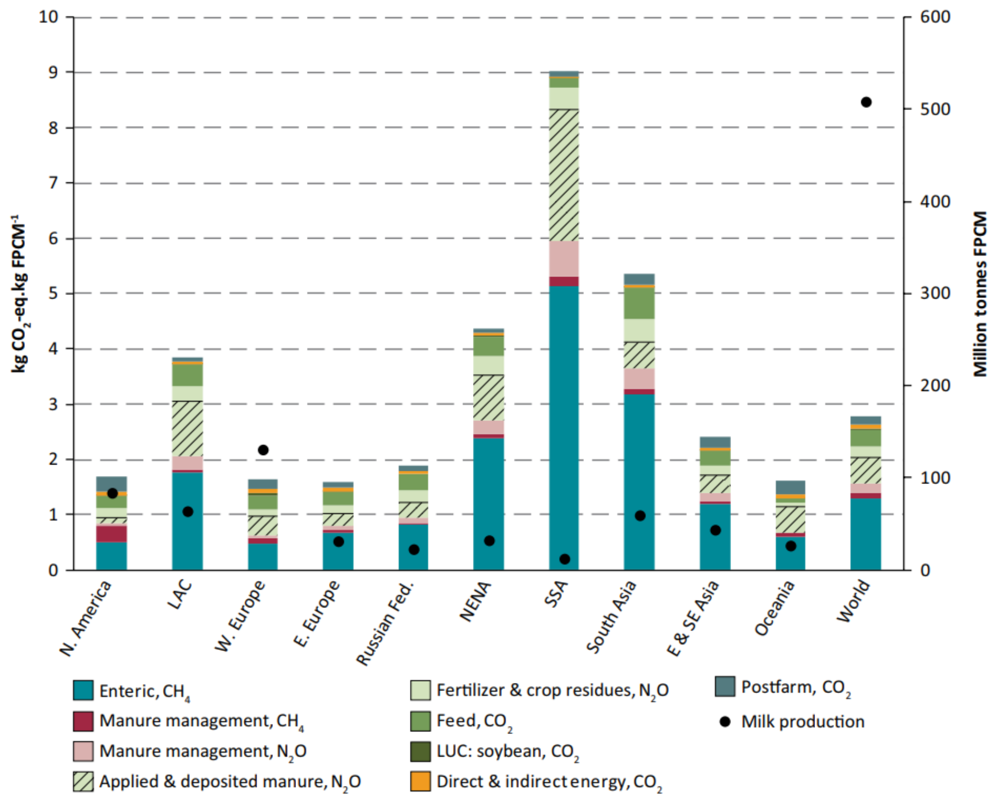


**Figure 2.5** Regional variation in milk production and greenhouse gas emission intensities (Gerber *et al.*, 2013a; LAC, Latin America; NENA, Near East and North Africa; SSA, Sub-Saharan Africa).

Few studies consider the 'global' boundary when reporting dairy GHG emissions. Generally, a proportion of the farm's GHG emissions are allocated to meat production, without taking into consideration the 'credit' of substituting a proportion of the beef market with dairy beef meat. Zehetmeier *et al.* (2014b) compared two 'boundaries' based on northern Europe dairy farm systems. The first boundary was referred to as 'all GHGs to milk' system, with beef meat confined to cull cows as the surplus calves were assumed to be sold to fattening beef farms, with the GHG emissions burden transferred elsewhere. The second boundary was referred to as 'systems expansion' system, where the boundary was expanded to include the fattening of surplus dairy calves, thus the emissions associated with the fattening of calves added to the dairy farm profile. Concurrently, an 'avoided-burden GHG credit' was also added to the dairy farm profile, representing the amount of GHG emissions that would have been emitted had the beef been produced in an alternative suckler-cow system. Three levels of cow milk production were examined in addition to two levels of uncertainty/variability. Considering only the variation in milk production, the EI for the 'all GHGs to milk' varied between 0.92 and 1.37 kg $CO_2$e/kg milk. When the systems expansion boundary combined with GHG emissions burden associated with the fattening of calves was added to the farm profile, EI varied between 1.35 and 1.65 kg $CO_2$e/kg milk. However, the credit for replacing meat in the suckler-cow beef system was substantive, at between 0.88 and 1.41 kg $CO_2$e/kg milk. The overall result was that the EI for the 6,000 kg milk/cow farm was reduced almost six-fold from 1.37 to 0.24 kg $CO_2$e/kg milk. For the 10,000 kg milk/cow farm, EI was almost halved from 0.92 to 0.47 kg $CO_2$e/kg milk.

The consideration of boundary is not isolated to the dairy industry. Browne *et al.* (2011) modelled the GHG emissions of a range of agricultural enterprises, including a comparison between an Angus cow-calf self-replacing system and a Hereford steer enterprise, both located in the same region of Victoria, Australia. There was little difference in the EI/ha between the two systems, at 5.12 and 4.81 t $CO_2$e/ha for the cow/calf and steer enterprises, respectively, most likely due to comparative stocking rate equivalents. However, on a per unit of product, there was a substantive difference in EI at 22.4 versus 6.3 t $CO_2$e/t product for the cow-calf and steer enterprises, respectively. The authors concede that for equitable comparison with self-replacing systems, emissions from replacement stock needs including, given the steers did not enter the 'farm boundary' until 18 months of age (Browne *et al.*, 2011). For dairy farms, replacement stock are often agisted off-farm post weaning and do not return to the dairy farm until just prior to calving for the first time. It is important that the emissions emitted from these animals are included in the dairy farm footprint and not attributed to the farm where they are agisted.

### *2.4.5. Allocation method*

Milk is the primary product of dairy farms. Meat, from culled cows and surplus calves raised for meat production, is a substantial co-product and thus needs consideration when allocating GHG emissions. Other possible and more minor co-products include hides, manure and energy from biogas production. The overall GHG emission impact should be partitioned among the various co-products. However, the handling of co-products is one of the most debated and unresolved issues in estimating dairy GHG emissions, since the allocation factors strongly affect the results (Nguyen *et al.*, 2013; Baldini *et al.*, 2017). Ideally, allocation should be avoided, either by dividing the process into sub-processes where emissions can be allocated to a single output or by systems expansion (Baldini *et al.*, 2017). However, dividing milk production into sub-processes is not possible (Flysjö *et al.*, 2011a).

Where allocation is unavoidable, it should be based on the biological relationship between products. The IDF (2010) recommend a biological method, centred on feed energy utilisation and quantified the energy required by the cow to produce a kg of milk or meat. In contrast, FAO (2010) highlighted the primary function of dairy farms is to provide humans with protein and thus proposed a protein allocation that enables a direct comparison with other protein sources. Where no other relationship can be identified, consensus is that allocation be based on economic value or the mass of the different products, even though these two methods have disadvantages (ISO 2006; IDF 2010).

Flysjö *et al.* (2011a) examined the effect of a range of allocation methods on the EI of milk production for two case study farms in New Zealand and Sweden. In both countries, systems expansion, defined as dairy meat replacing beef meat, resulted in a lower C footprint (kg $CO_2$e/kg energy-corrected milk (ECM)) compared with all other allocation methods. The systems expansion allocation method resulted in a 37% reduction in EI compared to zero allocation. Physical causality, based on the IDF (2010) guidelines, reduced EI by approximately 15%. All other allocation methods reduced EI between 2 and 12%, depending on the method and country examined. Similarly, Casey and Holden (2005b) compared the EI of milk production based on three allocation methods, finding that allocating all GHG emissions to milk resulted in 1.50 kg $CO_2$e/kg ECM, compared to 1.45 and 1.30 kg $CO_2$e/kg ECM for mass and economic allocation, respectively (Table 2.1). These are examples of reduction in EI that needs to be understood and taken into consideration when comparing results between studies.

### *2.4.6. Functional unit*

Most studies report the EI of milk production using one, occasionally two, functional units (FU); namely EI per unit of milk or per unit of land. Even the choice of equation to standardise milk is not well established. Generally the FU found in literature is either ECM or FPCM, however, there is variation in the formulae with either FU, resulting in potential variation in the results (Baldini *et al.*, 2017). The GHG emissions of production may vary depending on the FU implemented.

Ross *et al.* (2017) examined the effect of different FUs on EI based on seven years of data from four Scottish dairy systems; high forage or low forage intakes combined with either average UK genetics or top 5% of UK potential genetics. A range of FU's were explored including a newly proposed FU, GHG emissions per unit of milk yield produced per ha of total land use (kg $CO_2$e/t ECM.ha). This new FU was incorporated into the study to display the positive or negative outcome of trade-offs between production and land efficiencies, in which improvements in EI in one FU may be accompanied by deteriorations using another FU, thus adhering to a more standardised output/input measure of efficiency (Ross *et al.*, 2017). The authors found a significant ($P < 0.001$) difference in EI between each of the four dairy systems, varying between 0.83 and 1.10 kg $CO_2$e/kg ECM for the low forage/top genetics and high forage/average genetics dairy systems, respectively. When comparing the results using the new 'ECM/ha land' FU, there was only a significant difference between the two forages systems. The EI for the low forage/average genetics and low forage/top genetics systems were 14.9 and 13.4 kg $CO_2$e/t ECM.ha, respectively, significantly ($P < 0.001$) lower than 21.4 and 20.5 kg $CO_2$e/t ECM.ha for the high forage/average genetics and high forage/top genetics , respectively, illustrating the benefits of a low forage/high concentrate diet in reducing EI (Ross *et* *al.*, 2017). The choice of FU can have a bearing on whether there is a significant difference in EI between systems or mitigation options.

For a meaningful comparison of the environmental burden of different food products, a better comparison may be a FU based on the primary function of the product. For dairy, this is protein, and to a lesser extent, minerals. Vergé *et al.* (2013) examined the C footprint of 11 dairy products across all regions of Canada. There was variation in the C footprint within individual products between regions. The EI of cheese was 4.2 and 5.7 kg $CO_2$e/kg product in British Columbia and the Atlantic providences, respectively. Some of this variation could be explained by the large variation in EF for electricity between these two regions, at 0.046 and 0.593 kg $CO_2$e/kWh, respectively. The authors found that while the EI for cream and cheese was 2 and 5 kg $CO_2$e/kg product, respectively, the order reversed when compared on a per protein basis, with cheese having a much lower EI than cream, at 22 and 83 kg $CO_2$e/kg protein, respectively (Vergé *et al.*, 2013).

## 2.5 Mitigation strategies to reduce enteric methane emissions

With enteric $CH_4$ being the single largest GHG emission source from the dairy industry, there have been numerous reviews of mitigation options to reduce enteric $CH_4$ emissions (*e.g.* Beauchemin *et al.*, 2008; Eugène *et al.*, 2008; Eckard *et al.*, 2010; Hristov *et al.*, 2013a; Knapp *et al.*, 2014; Wanapat *et al.*, 2015; Moate *et al.*, 2016). These options have been broadly grouped into options such as animal manipulation, dietary manipulation including improving feed management, the use of plant secondary compounds or dietary supplements, and manipulating methanogens in the rumen to reduce enteric $CH_4$ (Figure 2.6; adapted from Eckard *et al.*, 2010; Hristov *et al.*, 2013b).

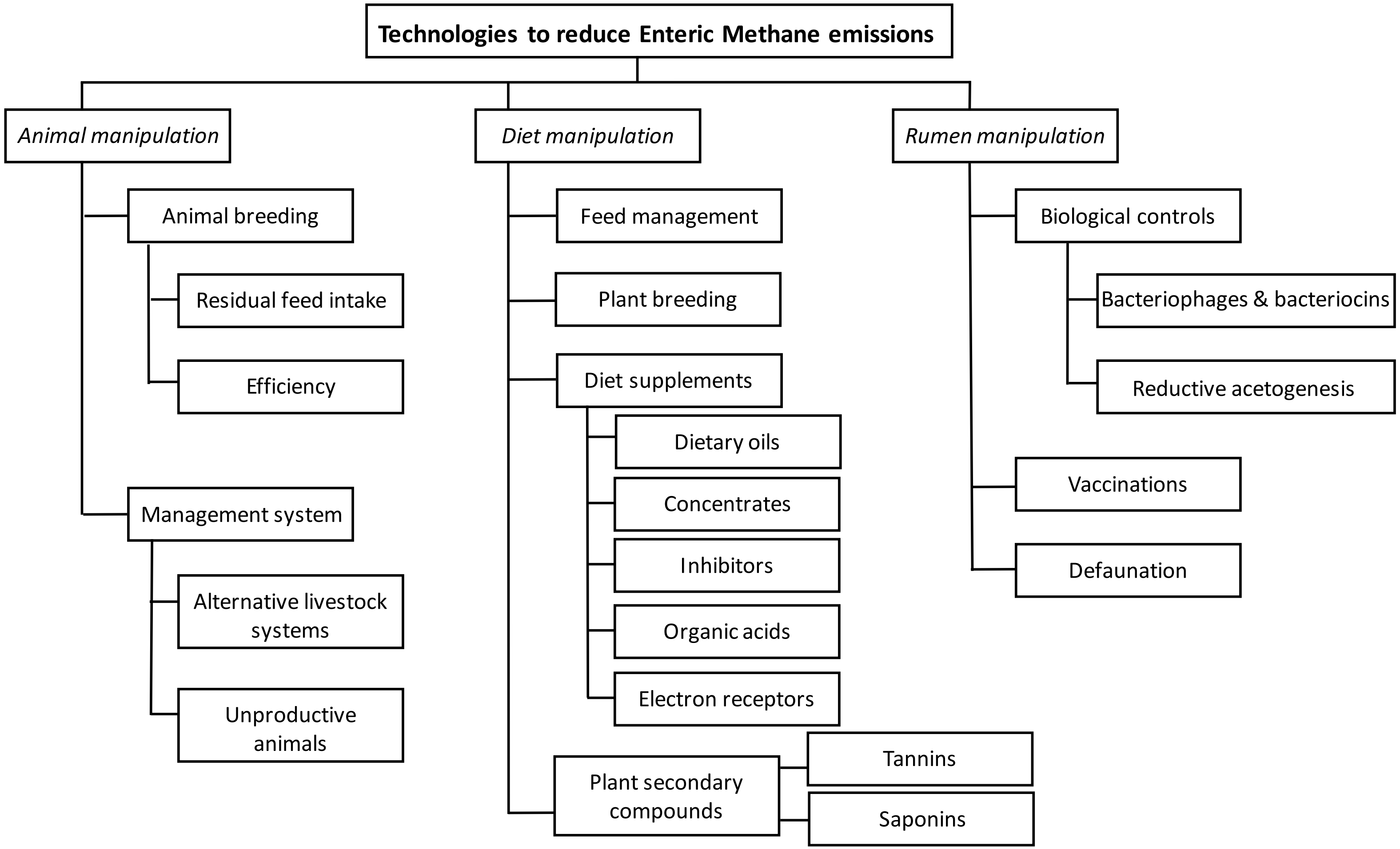


**Figure 2.6** A summary of existing strategies for the mitigation of enteric methane from ruminants (adapted from Eckard *et al.*, 2010; Hristov *et al.*, 2013b).

### *2.5.1. Animal manipulation*

The amount of $CH_4$ emitted by livestock is determined by the size of the animal, the quantity of feed intake (in part driven by productivity) and the quality of the feed (Blaxter and Clapperton, 1965; van Gastelen *et al.*, 2015; Morrison *et al.*, 2017). Improved milk production is a key to reducing the EI of milk production through dilution of the GHG emissions from the maintenance component of the animal. Production efficiency can be improved by genetic selection and management practices that address not only nutrition and feeding, but also reproduction, heat stress tolerance, disease incidence, culling rates, and heifer replacement programs (Knapp *et al.*, 2014).

Moate *et al.* (2016) compared the enteric $CH_4$ footprint of the Australian dairy industry for the years 1980 to 2010. They estimated that in the 1980s, the Australian dairy industry produced approximately 5.4 billion kg of milk from 1.88 million cows producing 2,900 kg/cow.lactation. By 2010, this had increased to approximately 8.9 billion kg of milk from 1.6 million cows producing 5,650 kg/cow.lactation. While milk production increased by 64%, $CH_4$ emissions only increased by 19%, thus the EI of milk production reduced from 33.6 g $CH_4$/kg milk to 23.9 g $CH_4$/kg milk, representing a 29% decline over the three decades. A similar result was found by Dyer *et al.* (2008) when examining the Canadian dairy industry, with EI reducing from 1.22 to 0.91 kg $CO_2$e/kg milk between 1981 and 2001. The Canadian national dairy cattle population (cows and heifers) declined by 37%, milk production (kg milk/cow.annum) increased by nearly 50% and total GHG emissions declined by 33%, resulted in a reduction in EI from 1.22 to 0.91 kg $CO_2$e/kg milk between 1981 and 2001. There is an upper limit to reducing EI by improved milk production efficiency and given that improving milk production per cow is not considered a key determinant of business success in grass fed systems (Bell *et al.*, 2013a; Bell and Wilson, 2018), other mitigation strategies will also need to be considered and implemented to further reduce the EI of milk production.

Intake variation from the predicted mean for individuals of a similar size and level of production has been termed residual feed intake, with efficient animals requiring less feed than their counterparts to produce the same level of product, lowering their net GHG emissions and thus EI of production (Waghorn and Hegarty, 2011). Connor *et al.* (2013) found variation in residual feed intake of 2.2 kg DMI/day between least and most efficient first-lactation heifers, increasing to a residual feed intake of 4.6 kg DMI/day between the least and most efficient cows in their 3+ lactation. There was no significant difference in milk production (kg ECM/head.day) between the high and low efficiency animals, thus the high efficiency animals were better at converting the lower DMI into milk production. Australian dairy farmers can select semen from bulls with greater weighting placed on improved feed efficiency (Australian Dairy Herd Improvement Scheme, 2015). Poor repeatability and high within-animal variation between residual feed intake and $CH_4$ emissions is present (Pinares-Patiño *et al.*, 2007; de Haas *et al.*, 2011; Lassen and Løvendahl, 2016), and therefore needs to be considered in combination with a suite of other mitigation options.

Researchers have also identified that there are variations between animals in terms of reduced $CH_4$ emissions with similar DMI (Pinares-Patiño *et al.*, 2007; Clark, 2013; Knapp *et al.*, 2014). Danielsson *et al.* (2017) examined 21 dairy cows identified as persistently low or high $CH_4$ emitters (g $CH_4$/day) over a 3-month period. Their findings were that while there was no significant difference in either DMI (kg DM/day) or milk production (kg ECM/cow.day) between the two groups, daily $CH_4$ emissions were significantly ($P < 0.001$) different, resulting in EIs of 8.3 and 9.7 g $CH_4$/kg ECM for the low and high emitting cows, respectively. The authors examined the differences in VFAs in the rumen fluid between the two groups, finding that the lower $CH_4$ emitting cows had a significantly ($P < 0.0001$) lower acetate + butyrate to propionate ratio compared to the higher $CH_4$ emitting animals (Danielsson *et al.*, 2017). Sequencing of rumen fluid found that the lower $CH_4$ emitting animals also had a 1.3 fold greater abundance of *Methanobrevibacter ruminantium* and 1.5 fold lower abundance of *Methanobrevibacter gottschalkii* relative to the higher $CH_4$ emitting animals (Danielsson *et al.*, 2017). The results of these authors affirms the link between methanogen populations, volatile fatty acid production and enteric $CH_4$ emissions (Hook *et al.*, 2010; Knapp *et al.* 2014; Leng, 2014; Danielsson *et al.*, 2017). Breeding animals with lower $CH_4$ production, without compromising on intake and production, reduces net emissions and thus would be a win:win mitigation strategy for the environment and for the farmer.

Many research mitigation options focus on the largest source of GHG emissions, the milking cow. Estimation of dairy GHG emissions by Christie *et al.* (2012) found that enteric $CH_4$ from replacement heifers averaged around 10% across the 41 Australian dairy farms. However, for some farms, emissions from replacements was as high as 15-20% of total farm GHG emissions. Given the substantive GHG emissions associated with raising heifers from birth until they enter the milking herd, this is an area of research that needed consideration.

Christie *et al.* (2016; Chapter 6), using a modelling approach, examined the effect of diet quality on the time required for grazing heifers to reach a LW suitable for mating. This is a widespread issue for subtropical Australia, where the typical time between birth and 1st lactation calving is can be as high as 34 months, primarily because of heifers grazing lower quality pastures compared to their southern counterparts, which typically calve for the first time at 24 months of age. The time required and enteric $CH_4$ emissions produced in raising dairy heifers to a target LW for mating was examined using a dynamic approach that facilitated the examination of the effect of climate variability on pasture availability, nutritive value and associated GHG emissions. Modelling 44 years of climate data showed that mean age at first mating was reduced from 22 to 17 months, with cumulative $CH_4$ emissions reduced from a mean of 1.2 to 0.7 t $CO_2$e/head from weaning to mating when the diet metabolisable energy (ME) was increased from 9.5 to 10.9 MJ/kg DM, respectively. This strategy of increasing daily LWG to promote earlier mating and earlier calving would result in a reduction in the animal's lifetime EI of milk production.

Knapp *et al.* (2014) illustrated that for a herd where heifers calved at 24 months of age, the number of replacements required per 100 cows could be reduced from 71 to 59 head if the percentage of herd culled each year was decreased from 30 to 25%. Enteric $CH_4$ emissions from this cohort of heifers were reduced from 24 to 21% of the whole herd's emissions. Based on a milking herd of 300 cows and assuming 1 t $CO_2$e/heifer based on results by Christie *et al.* (2016), this could equate to a saving in enteric $CH_4$ of 36 t $CO_2$e/annum. This is a small reduction in GHG emissions when considering the overall farm's GHG emissions, but could be used in combination with a range of other mitigation strategies to reduce the overall C footprint.

Browne *et al.* (2015) examined the effect of extending the traditional lactation length from 300 days (*i.e.* calving every 12 months), the traditional length for Australian dairy farms, to 480 days (*i.e.* calving every 18 months). Results were modelled over six years, with six lactations for the annual calving cows and four lactations for the extended lactation cows. Total GHG emissions, averaged over the six years, was 100 t $CO_2$e/annum lower for the extended lactation herd. Daily milk production (kg FPCM/cow.day) was similar between the traditional and extended lactation herds, but the longer period of lactation for the latter resulted in greater annual milk production (kg FPCM/cow.annum). Both factors contributed to a reduction in EI for the extended lactation herd by 6%. Although this strategy only had a fractional reduction in emissions, this can be viewed as a potential win:win strategy with improvements in profitability also identified (Browne *et al.*, 2015). Therefore, extended lactation as a mitigation option needs to be brought to the attention of the Australian dairy industry to lift overall performance with respect to both GHG emissions and farm profitability.

### *2.5.2. Diet manipulation*

Increasing the proportion of concentrate in the diet has been proven to lower enteric $CH_4$ emissions per unit of DMI and per unit of product if production remains the same or increases (Johnson and Johnson, 1995, Boadi *et al.*, 2004). Concentrate feeding also enhances the proportion of propionate relative to acetate, thus decreasing the amount of $H_2$ available for conversion to $CH_4$, a key ratio associated with reducing enteric $CH_4$ emissions (Hristov *et al.*, 2013a; Knapp *et al.*, 2014). The type of concentrate can also have an effect on $CH_4$ emissions. Most international studies assessing the effect of concentrate feeding have focused on maize grain and soybean meal (Yan *et al.*, 2010; Powell *et al.*, 2017). In an Australian study comparing maize to wheat and barley, Moate *et al.* (2017) concluded that $CH_4$ emissions were significantly ($P < 0.001$) reduced with the wheat diet compared to maize and barley diets, due to the higher starch degradation in the rumen (Moharrery *et al.*, 2014). The authors noted that while total milk production (kg milk/cow.day) did not alter between grains, milk fat depression was significant ($P < 0.01$) with the wheat diet (Moate *et al.*, 2017). Thus, the benefits of a reduction in GHG emissions needs to be considered in relation to changes in income from lower composition milk, especially if no other source of increased profit can be realised with the lowering of GHG emissions.

Pacheco *et al.* (2014) reviewed research undertaken in New Zealand where sheep were fed a variety of fresh forages in respiration chambers to measure enteric $CH_4$

emissions. Enteric $CH_4$ yield (g $CH_4$/kg DMI) was lower in animals fed forage rape (*Brassica napus*) compared to perennial ryegrass (*Lolium perenne*). The variation in $CH_4$ yield was explained by the differences in the ratio of readily fermentable carbohydrate to structural carbohydrate between the two forages. The rumen microbial population present in the lambs fed forage rape was comparative to microbial populations in grain fed animals (Pacheco *et al.*, 2014). Feeding forage rape to lambs has also been shown to increase LWG, relative to perennial ryegrass pastures (Hopkins *et al.*, 1995; Pacheco *et al.*, 2014), resulting in a combined 38-46% reduction in EI (Pacheco *et al.*, 2014).

Increasing forage digestibility will generally reduce $CH_4$ emissions from rumen fermentation (van Laar *et al.*, 2002; Beauchemin *et al.*, 2008). Guyader *et al.* (2017) examined the GHG emission implications of feeding either a maize or barley-based diet, across three levels of diet DM digestibility (DMD) to lactating dairy cattle. The EI was 1.08 kg $CO_2$e/kg FPCM for the maize diet compared to 1.24 kg $CO_2$e/kg FPCM for the barley diet when DMDs were 69 and 65% for the maize and barley diets, respectively, illustrating differences between diets. Increasing the maize diet from low to medium DMD (*i.e.* from 64 to 66%) reduced EI from 1.30 to 1.18 kg $CO_2$e/kg FPCM. Increasing the barley diet from low to medium DMD (i*.e.* from 60 to 63%) reduced EI from 1.52 to 1.37 kg $CO_2$e/kg FPCM, respectively.

Brask *et al.* (2013) examined the effect of feeding early cut grass silage, late cut grass silage or maize silage on dairy cattle $CH_4$ emissions. While there was no significant difference in $CH_4$ emissions (kg $CH_4$/day) between the three feeds, the percentage of gross energy emitted as $CH_4$ was 5.6% with the maize silage, which was significantly ($P < 0.01$) lower than 6.4% and 6.9% for the early and late cut grass silages, respectively. These results highlight the importance of choice of metrics when comparing treatments. There was no significant difference in daily $CH_4$ emissions (kg $CH_4$/day) between the three silages, although there was a significant ($P < 0.05$) difference when reporting $CH_4$ emissions as a percentage of gross energy. In addition, while not analysed in the study, the lower crude protein (CP) concentration of the maize silage, relative to the grass silage, may have also resulted in lower $N_2O$ emissions from the animal excreta for the maize silage-fed cows, thus highlighting the importance of a whole-of-system analysis of emissions, as opposed to a single GHG source.

Jonker *et al.* (2018) compared the enteric $CH_4$ emissions from sheep fed either a conventional diploid, a high water-soluble carbohydrate (WSC) diploid or a tetraploid perennial ryegrass over three seasons. Averaged over the three feeding seasons, $CH_4$ emissions (g $CH_4$/kg DMI) were significantly ($P < 0.001$) greater with the conventional diploid ryegrass pasture compared to the high WSC and tetraploid ryegrass pastures, although this was not consistent between seasons, indicating a seasonal by cultivar interaction ($P < 0.001$). Overall DMI for the sheep consuming the tetraploid ryegrass was significantly ($P < 0.05$) lower than the other two diploids, which partly explained the difference in $CH_4$/kg DMI between the conventional diploid and tetraploid. However, there was no significant difference in LWG/ha between the three ryegrass cultivars across all three seasons (Cosgrove *et al.*, 2015).

There is a large body of evidence that lipids suppress $CH_4$ emissions (Eugène *et al.*, 2008; Moate *et al.*, 2011; Jayasundara *et al.*, 2016). Grainger and Beauchemin (2011) analysed 27 studies and concluded that, within the practical feeding rate of < 8% dietary fat, a 10g/kg increase in dietary fat reduced $CH_4$ emissions by 1g $CH_4$/kg DMI (5% reduction) in cattle. Moate *et al.* (2014) found that feeding grape marc, a by-product from the wine industry that is high in crude fat, in either a dried form (diet mean of 5.7% crude fat) or an ensiled form (diet mean of 5.2% crude fat), reduced daily $CH_4$ emissions (g $CH_4$/cow.day) by 20 and 17%, respectively, relative to the 2.6% crude fat control diet. There was no significant ($P < 0.05$) difference in milk production (kg milk/cow.day) between the control diet and the two grape marc diets. However, when milk production was corrected for fat and protein production (kg ECM/cow.day), the two grape marc diets resulted in a significant ($P < 0.001$) depression in milk fat content, relative to the control diet. Thus, the reduction in $CH_4$ emissions may need to be balanced against a potential reduction in income from the milk derived from grape marc-fed cows if this is to be considered a suitable mitigation option.

Condensed tannins (CT) have often (Gerber *et al.*, 2013b), but not always, been shown to reduce $CH_4$ emissions (Beauchemin *et al.*, 2007). Woodward *et al.* (2002) examined the $CH_4$ emissions of milking cows fed either a diet of perennial ryegrass or the CT-rich pasture species sulla (*Hedysarum coronarium*). Relative to the perennial ryegrass-grazing cows, those grazing sulla substantially increased their DMI (kg DM/cow.day) and milk yield (kg milk/cow.day), reducing $CH_4$ production (g $CH_4$/kg DMI) by 21%. Beauchemin *et al.* (2007) examined the feeding of a CT extract from quebracho trees (*Schinopsis quebracho- colorado*) to beef cattle at rates of 1 and 2% of dietary DM. The authors found there was no effect of the CT on $CH_4$ emissions (g $CH_4$/day, g $CH_4$/kg DMI or % gross energy intake), although they found evidence of the protein-binding effect of the CT, thus having $N_2O$ emission implications (Beauchemin *et al*., 2007).

Ludemann *et al.* (2016) examined the net mitigation potential of whole cottonseed (WCS) when used as a fossil-fuel substitute (biodiesel) or as a feed source to cattle. Their conclusions were that the GHG abatement of converting WCS into biodiesel (and associated displacement of diesel GHG) was greater than the reduction in enteric $CH_4$ when fed to cattle. Feeding WCS to stock was estimated to result in GHG abatement, assuming the distances required to transport the supplement was no greater than 380 km from the processing facility. This negates the benefit of transporting cottonseed from northern New South Wales to dairying regions in Victoria.

In a similar study, Williams *et al.* (2014) compared feeding a wheat-based diet with feeding WCS to dairy cattle over summer. On-farm GHG emissions (t $CO_2$e) were reduced by 1% by feeding the WCS relative to the wheat-based diet. However, pre-farm gate emissions increased by 3% due to the emissions associated with transporting the WCS from the processing plant to the farm. The net result was that total GHG emissions, when including production and transport, were similar between the two feeding regimes.

Kinley *et al.* (2016) explored the inclusion of the red macroalgae *Asparagopsis taxiformis* with Rhodes Grass (*Chloris qayana*) substrate on enteric $CH_4$ emissions, via an *in vitro* assessment over 72 hours. Five *Asparagopsis* dose rates were examined between 0.5 and 10% organic matter (OM) basis, finding enteric $CH_4$ emissions (mL $CH_4$/g OM) were virtually eliminated at doses $\geq$ 2% OM basis. There was also no negative impact on substrate digestibility with the macroalgae inclusion at doses $\leq$ 5% OM. Li *et al.* (2018) explored the potency of *Asparagopsis* to reduce enteric $CH_4$ emissions over a 72-day period using adult wether sheep. The incorporation of *Asparagopsis*, at an equivalent of 3% OM basis, reduced enteric $CH_4$ production (g $CH_4$/day) and $CH_4$ yield (g $CH_4$/kg DMI) by 80% (Li *et al.*, 2018). Just as important, there was no indication of adaption of the methanogens to the algae over the 72-day period (Li *et al.*, 2018). However, there is general reluctance to feeding *Asparagopsis* to livestock, as the active ingredient compound bromoform has a similar chemical potency to bromochloromethane, which is linked to ozone depletion (Kinley *et al.*, 2016). It remains unclear whether this will prove to be a practical and acceptable method for reducing enteric $CH_4$ emissions.

### *2.5.3. Rumen manipulation*

Ruminant $CH_4$ emissions can also be manipulated by changing the rumen microbial population through a range of actions, including alternative H sinks such as propionate and butyrate enhancers, plant secondary metabolites such as tannins, saponins and essential oils, inhibitors such as ionophores and bacteriocins along with anti-methanogen vaccines (Hristov *et al.*, 2013a; Patra *et al.*, 2017). Monensin has been the most studied ionophore and is routinely used in North America to improve feed efficiency in feedlot and pasture-grazed beef and dairy cattle. However, the effect of reducing $CH_4$ emissions appears to be inconsistent (Hristov *et al.*, 2013a), with some studies finding a reduction in $CH_4$ emissions (Sauer *et al.*, 1998), and others not (Benchaar, 2016). There are regulatory restrictions in using ionophores in some countries and the efficacy for animals grazing pastures have not been consistent (Hristov *et al.*, 2013a).

The symbiotic and cross-feeding relationship between ruminal protozoa producing large quantities of $H_2$ and methanogenic archaea removing this $H_2$ has been well established (Hristov *et al.*, 2013a; Belanche *et al.* 2014). In a meta-analysis of *in vivo* experiments by Morgavi *et al.* (2010), $CH_4$ reduction associated with the defaunation of protozoa was about 10%, however the data was highly variable, with some treatments resulting in an increase in $CH_4$ emissions. Protozoa play an important role in fibre and OM digestion in the rumen, so defaunation has the potential to impact digestibility, productivity and milk fat concentration. In addition, a reduction in protozoa could result in an increase in the population of bacteria-associated methanogens, counteracting a potential reduction in $CH_4$ emissions (Eugène *et al.*, 2004; Hristov *et al.*, 2013a).

Bacteriocins are proteins or peptides produced by bacteria and are believed to moderate rumen fermentation, leading to increased propionate production, therefore reducing $CH_4$ production (Parta *et al.*, 2017). While studies have shown promising results, with suppression of up to 50% *in vitro* (Lee *et al.*, 2002; Sar *et al.*, 2005), there appears to have been no follow-up research to examine their efficacy *in vivo* or to determine the cost of implementation (Patra *et al.*, 2017). Nitrates have been shown to decrease $CH_4$ production both *in vitro* and *in vivo* (Olijhoek *et al.*, 2016; Patra *et al.*, 2017). However, the suitability of $NO_3$ supplementation for dairy cattle may be considered low, given that most intensively managed dairy pastures already contain adequate concentrations of N (Whitehead, 1995; Reeves *et al.*, 1996; Lawson *et al.*, 2017), coupled with the major concerns regarding its potential toxicity (Jayasundara *et al.*, 2016).

The synthetic compound 3-Nitrooxypropanol (3NOP) has been shown to have anti-methanogenic properties. Haisan *et al.* (2017) examined the effect of feeding 3NOP, at rates of either 1.25 or 2.5 g 3NOP/day, to mid lactation Holsteins over a 28-day period. While the feeding of 3NOP had no significant effect on intake (kg DMI/cow.day) or milk yield (kg ECM/cow.day), enteric $CH_4$ emissions (g $CH_4$/kg ECM) were significantly ($P < 0.001$) reduced by 27 and 42% with the low and high 3NOP rates, respectively, compared to the control diet. In another study, Hristov *et al.* (2015) examined the effect of feeding three rates of 3NOP (40, 60 and 80 mg/kg DM equivalent to between 1.1 and 2.2 g 3NOP/day) to early lactation Holstein cows over a 12-week period. Enteric $CH_4$ emissions (g $CH_4$/kg DMI and g $CH_4$/kg ECM) were reduced by 30% over the 12-week period, relative to the control diet. At the end of the experiment, $CH_4$ emissions ($CH_4$/kg DMI) was on average 25% lower with feeding 3NOP, indicating a decrease in efficacy over time.

Haisan *et al.* (2017) and Hristov *et al.* (2015) postulated that due to the high water-solubility of 3NOP and the rapid turnover of rumen contents in lactating dairy cattle, continuous consumption through a total mixed ration feeding system, typical of dairy systems seen in North America and Europe, would allow for a more continuous effect of reducing $CH_4$ emissions. In Australia, these rates of 3NOP could be delivered via supplementary feeding during milking twice a day or via a bolus delivered into the rumen to allow continual supply. Exploration of the suitability and efficacy of 3NOP delivery via these methods therefore needs to be undertaken to ascertain if this is a potential mitigation for pasture-based grazing systems. Further research also needs to ascertain the carry-over of the compound into milk and food safety concerns when consumed by humans (Jayanegara *et al.*, 2018).

Vaccines have been explored to generate a salivary antibody response that delivers neutralising antibodies to the rumen. Wright *et al.* (2004) found that a vaccine, based on whole cells, had mixed effects on methanogen populations or $CH_4$ emissions in sheep, with follow-up work using New Zealand and Australian methanogen strains proving unsuccessful (Clark *et al.*, 2005). Using a different approach, based on using cell fractions as opposed to whole cells, Wedlock *et al.* (2010) demonstrated the stimulation of antibodies to suppress methanogen growth and $CH_4$ emissions *in vitro*. This modified approach needs confirmation *in vivo* (Clark, 2013), with Patra *et al.* (2017) suggesting that most antibodies circulate in the blood of the host, with only a tiny amount entering the rumen via saliva. Thus, vaccination may prove infeasible in the short to medium term.

Great promise has been placed on the role of genome sequencing research to identify the structure, function and metabolic diversity of ruminal methanogen groups in providing insights into mitigation options to reduce enteric $CH_4$ emissions (Attwood *et al.*, 2011; Zhou *et al.*, 2011; Kittelmann *et al.* 2014; Henderson *et al.*, 2015). While research groups throughout the world are working on developing natural and synthetic compounds that directly inhibit methanogenesis, findings to date have suggested large discrepancies exist in the efficacy, with some reporting adverse impacts on feed digestion and therefore production (Hristov *et al.*, 2013a; Patra *et al.*, 2017). Until there is consistency of response, with little to no adverse effect on overall animal performance, in addition to a cost-benefit assessment of the benefit of such options, rumen manipulation to reduce $CH_4$ emissions is unlikely to become the 'silver bullet' solution, especially if the compound is synthetically derived and thus needs to overcome regulatory and consumer barriers to adoption.

### 2.6 Mitigation strategies to reduce waste methane emissions

Most of the $CH_4$ emissions from manure is produced under anaerobic conditions during storage. Strategies typically viable for farms in Europe and North America include the use of anaerobic digesters or capturing $CH_4$ to be flared, produce heat or electricity. However, these options are unlikely to become economically viable for most Australian dairy farms due to the small amount of manure captured on hard surfaces (10 to 15% of total depositions), with the remaining manure deposited onto pastures while grazing. Large total mixed ration dairies are the most appropriate sites for installing anaerobic digesters. However, there are few total mixed ration dairies in Australia (Edgerton, 2009; Dairy Australia, 2015a) and thus unlikely to be a substantial method of reducing waste emissions. Where it is necessary to store waste, such as pond/lagoon systems, reducing storage time is imperative to reduce the period of anaerobic conditions conducive to $CH_4$ loss (Gerber *et al.*, 2013a).

The most important mitigation strategy to reduce waste $CH_4$ emissions is improving the digestibility of the diet to reduce the production of manure. Reductions in manure production can also be achieved through many of the herd management options listed in the enteric $CH_4$ strategies above, such as improving milk production efficiency to dilute total emissions per unit of milk, reducing age to first calving and improved animal health (Montes *et al.*, 2013).

### 2.7 Mitigation strategies to reduce nitrous oxide emissions

There have been numerous reviews of mitigation options to reduce $N_2O$ emissions (*e.g.* Dalal *et al.*, 2003; de Klein and Eckard, 2008; Luo *et al.*, 2010; Montes *et al.*, 2013; Li *et al.*, 2015; Di and Cameron, 2016, Jayasundara *et al.*, 2016). Options to reduce $N_2O$ emissions fall into two broad categories; improving the efficiency of N cycling in animal production and improvements in soil management (Figure 2.7; adapted from Eckard *et al.*, 2010).

Some mitigation options that are more relevant to housed animals, such as manure handling, storage and application to land are not discussed here, as the relevance of these to Australian dairy farming is minimal. However, for farms where herds are spending time on feedpads to receive a substantial component of their diet from supplementary feed (*i.e.* FS3 to FS5 farms), the amount of manure collected will increase. Under these circumstances, some additional strategies could become relevant into the future such as composting of collected manure or decreased storage time of effluent before spreading onto pastures or crops (Montes *et al.*, 2013; Aguirre-Villegas and Larson, 2017; Guest *et al.*, 2017). Animal and soil management options will remain the critical areas of intervention to reduce $N_2O$ emissions.

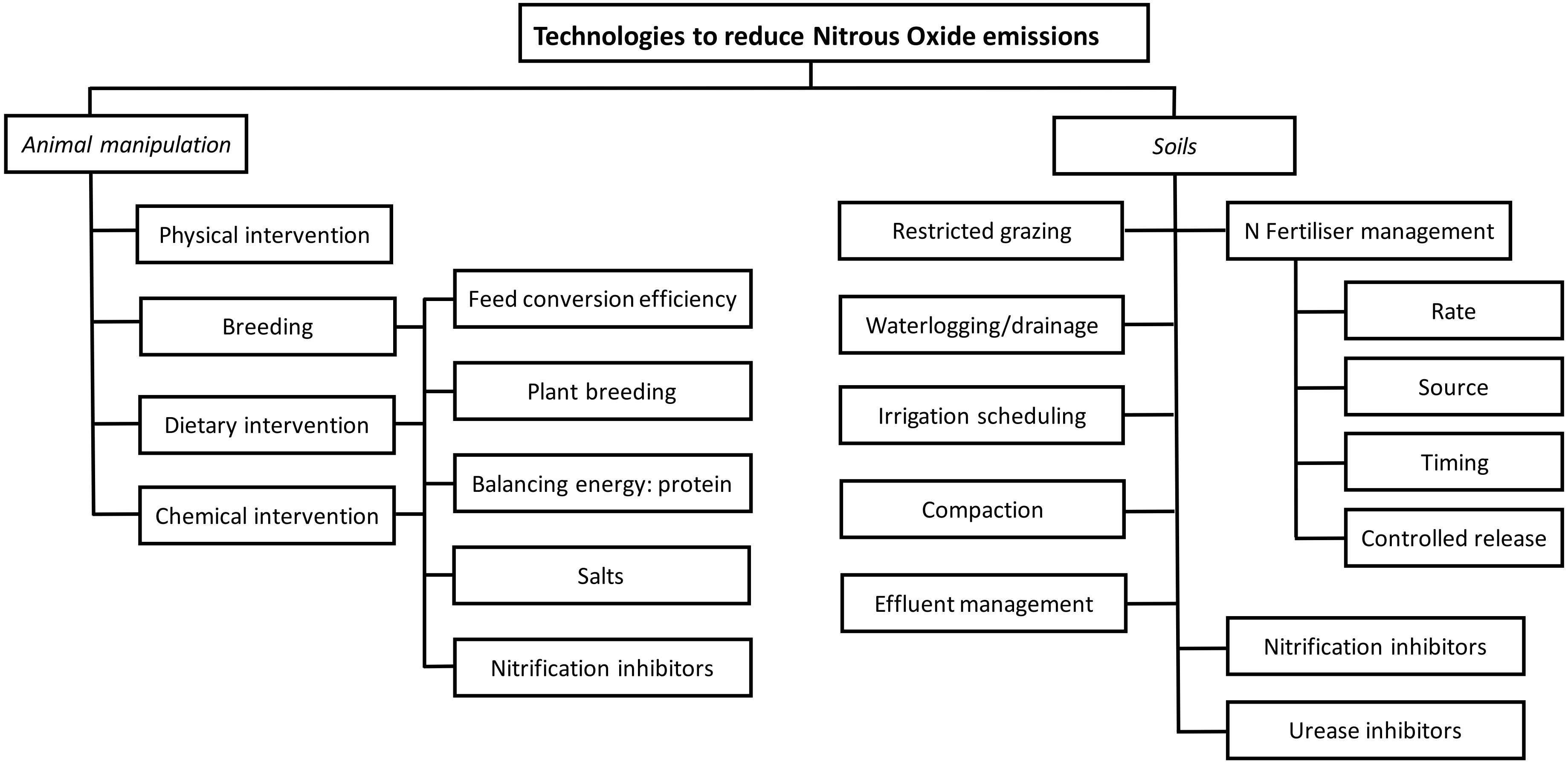


**Figure 2.7** A summary of existing strategies for the mitigation of nitrous oxide from ruminant production systems (adapted from Eckard *et al.*, 2010).

### *2.7.1. Animal management interventions*

In a manner already discussed for $CH_4$ mitigation, improving the efficiency of animal productivity will dilute $N_2O$ emissions. In addition, there are some mitigation strategies specifically targeted to reducing $N_2O$ emissions.

It is generally accepted that lactating dairy cows require a CP concentration of 16 to 18% in their diet for milk production. However, well fertilised pastures can have CP concentrations > 20% (Chapman *et al.*, 2012; Cullen *et al.*, 2017; Loaiza *et al.*, 2017) and as such, typically contain an excess of N relative to animal requirements. With <30% of N intake utilised for production, the remainder is excreted in dung and urine (Whitehead, 1995). Given that the concentration of N in dung remains relatively stable, the excess is excreted in urine, at rates far greater than the soil-plant system can effectively utilise (Whitehead, 1995; Eckard *et al.*, 2010).

Optimum grazing management, based on leaf stage, is critical to balancing the energy-to-protein ratio to minimize $N_2O$ emissions. Grazing pasture prior to reaching their optimum leaf stage for defoliation, results in increased CP concentration and decreased WSC concentration in the leaf, reducing the energy-to-protein ratio (Reeves *et al.*, 1996; Rawnsley *et al.*, 2002; Turner *et al.*, 2006; Lawson *et al.*, 2017), thus increasing N excretion by the animal (Miller *et al.*, 2001; Staerfl *et al.*, 2012). Alternatively, grazing pastures beyond their optimum leaf stage has been shown to increase fibre concentration (Abraham *et al.*, 2009; Loaiza *et al.*, 2017; Pembleton *et al.*, 2017), decreasing the digestibility of the pastures, thus increasing dung excreta and associated $CH_4$ emissions.

Misselbrook *et al.* (2005) found that dairy cows fed a 14% CP diet excreted 30% less urinary N (based on concentration and volume) than dairy cows consuming a 19% die. Urine contained 52% of total N output with the low CP diet compared to 68% with the high CP diet. Külling *et al.* (2001) compared the effect of three levels of CP concentration diets fed to dairy cattle (12.5, 15.0 and 17.5% CP) on total N excreted and subsequent $N_2O$ emissions from stored manure. Reducing the diet concentration from 17.5 to 15% CP decreased urinary excretion (g N/kg excreta) by 44%, with $N_2O$ losses (% of N loss) reduced by over 60%. Luo *et al.* (2008a) compared two farm systems; a control system with a lower stocking rate of 3.0 cows/ha without supplement versus a mitigation system with a higher stocking rate of 3.8 cows/ha and including maize silage supplementation during autumn and winter when pasture growth was slow. Including all additional direct and indirect GHG emissions associated with the low-protein maize supplement, emissions were 8.0 and 8.3 kg $N_2O$-N/ha for the control and maize supplement systems, respectively. However, the maize supplementation increased milk production (kg milk/ha.annum) by 33%, thus decreasing EI from 0.59 kg $N_2O$-N/t milk with the control system down to 0.46 kg $N_2O$-N/t milk with the maize supplementary system.

Christie *et al.* (2014; Chapter 5) implemented a modelling approach to evaluate two options for improving animal NUE and reduce $N_2O$ emissions for three dairying regions of south-eastern Australia. Sixteen scenarios were examined: four MN concentrations, representing 3.1%, 3.4%, 3.8% and 4.1% milk protein, combined with four SN concentrations, representing 1%, 2%, 3% and 4% N. Exploring these combinations across a range of spatial and temporal scales would be practically impossible with field experimentation, highlighting the role of modelling in exploring complex mitigation options. Supplementary intakes were purposely set high to maximise supplementary feed intakes and thus reduce the N concentration of the overall diet. Simulations were run for a 20-year period to incorporate the effects of weather variability on pasture production and supplementary feed requirements. When the N concentration (g N/kg DM) in supplementary feed was reduced from 4% to 1%, reducing mean diet N concentration from 4.1% to a mean of 2.4% across the three regions, $N_2O$ emissions declined by between 50 and 57%. Improving MN concentration to reduce $N_2O$ emissions was less pronounced, with declines of between 7 and 11%. This was an important finding for the Australian dairy industry, since manipulation of dietary N to better balance the energy to protein ratio would be more readily achievable than manipulation of N concentration in milk through genetic selection.

Condensed tannins have been shown to reduce $N_2O$ by forming complexes with protein, slowing the rate and extent of rumen protein degradation, reducing rumen $NH_3$ concentrations and thus reducing the excretion of urinary N (Griffiths *et al.*, 2013). Black wattle (*Acacia mearnsii*) has shown to partition N away from urine towards faeces. However, it was shown to have a negative impact on milk production when the diet was marginal for CP concentration (Grainger *et al.*, 2009). When dairy cattle were fed a powdered form of black wattle extract in combination with pastures with excess N concentration (> 20% CP), there was a significant ($P < 0.001$) decline in milk yield (kg milk/cow.day) as the CT concentration increased, although milk composition (fat, protein and lactose concentration) was not affected (Griffiths *et al.*, 2013). Faeces N concentration (% of DM) increased with increased rates of CT, suggesting portioning of dietary N away from urine. When the CT was incorporated into a pellet, the effect of the CT on milk yield (kg milk/cow.day) was no longer significant, indicating that the form of CT delivery may influence the GHG emissions reduction potential (Griffiths *et al.*, 2013).

### *2.7.2. Soil management interventions*

There are several interventions to minimise $N_2O$ losses through better soil management, such as fertiliser management, manure management or use of nitrification inhibitors (NI). The rate, source, type and timing of N fertiliser applications are important management factors affecting the efficiency of pasture growth responses, and thus the magnitude of N loss (de Klein and Eckard, 2008). Synchronising timing of fertiliser application with plant N demand is an important measure to improve NUE (Luo *et al.*, 2010). Using a modelling approach, Smith *et al.* (2018) compared NUEs (kg DM/kg N fertiliser above the zero N treatment) and N losses (kg N/ha.annum) when applying a consistent flat rate of N fertiliser monthly, as opposed to only applying according to plant demand, across five sites in Australia. The authors found that mean NUEs were improved by between 17 and 41%, depending on the FS and location, with N losses reduced by between 19 and 45% when applying N fertiliser according to plant demand as opposed to a monthly flat rate (Smith *et al.*, 2018).

Strategic application of manure to pastures under low soil-moisture status could potentially reduce $N_2O$ emissions by up to 96% (Luo *et al.*, 2010). Delaying effluent or manure applications after grazing events could further reduce $N_2O$ emissions by reducing the level of surplus mineral N in the soil (Luo *et al.*, 2010). Restricted grazing in New Zealand has shown to reduce direct ($N_2O$) and indirect ($NO_3$) losses (kg N/ha) by 57 and 41% for three autumn/winter periods (de Klein *et al.*, 2006). Luo *et al.* (2008b) compared winter grazing (control) to restricted grazing (6 hours per day over 90 days of winter). They found that overall farm emissions were 7.7 kg $N_2O$-N/ha with the control diet, declining to 7.0 kg $N_2O$-N/ha with restricted grazing, mostly driven by the lower direct $N_2O$ emissions associated with getting cows out of the paddock during winter.

Technologies employing NIs and urease inhibitors (UI) are effective mitigation alternatives to control N losses by acting on the N processes of urea hydrolysis and nitrification (Zaman *et al.*, 2008; Luo *et al.*, 2010). During a 3-month study in New Zealand (late spring/early summer), Zaman *et al.* (2008) examined the effect of the UI N-(n-Butyl)thiophosphoric triamide (NBPT; trade name Agrotain®) on N losses. The authors found that, relative to the urea only control treatment, incorporation of the UI reduced $NH_3$, $NO_3$ and $N_2O$ emissions by 45, 47 and 5%, respectively, coupled with an increase in pasture production of 17% (Zaman *et al.*, 2008). The authors speculated that the increase in pasture production was beyond what could be explained solely by the reductions in N loss and thus one possibility was that less energy was required by the plant to uptake $NH_4$ relative to $NO_3$ (Zaman *et al.*, 2008).

A review of studies from New Zealand by Di and Cameron (2016) found the mean efficacy of NIs with reductions from urine patch $NO_3$ leaching and $N_2O$ emissions in the order of 50 and 57%, respectively. In some of these NZ studies, pasture production increased by 20-25%, but was variable in others, which the authors suggest is not surprising in view that many factors can affect pasture growth, not just the availability of N in the soil (Di and Cameron, 2016). It is important to remember the efficacy of NIs are temperature and soil moisture dependant. Some studies in Australia resulted in no significant reduction in N loss coupled with no increase in pasture production with implementation of the NI (Dougherty *et al.*, 2016). Even where there were significant ($P < 0.05$) reduction in $N_2O$ losses, the amount of 'saved' N with the NI did not necessarily translate to additional pasture production (Kelly *et al.*, 2008; Suter *et al.*, 2016). If conditions were conducive to substantial increases in pasture production, consideration would need to be made towards the increase in stocking rate to consume this additional pasture as this would increase $CH_4$ emissions and therefore net farm GHG emissions.

Unfortunately, in 2012, traces of the NI dicyandiamide (DCD) was found in milk samples in New Zealand, resulting in the product being removed from the Australian and New Zealand marketplace (Astley, 2013). Although DCD has been used commercially for decades and is recognised as non-toxic, there currently is no declared Maximum Residue Limit under the *Codex Alimentarius* (international food safety standards), hence a default limit of zero residue applies. A process is currently underway to identify a threshold level of residues of compounds (like DCD and others) with a very low toxicology that could be introduced to the *Codex Alimentarius*, which would allow the re-introduction of DCD for commercial use. It is anticipated that the guidelines will be adopted as an international standard by July 2019 (Eckard and Clark, 2018), after which DCD may once again be a viable option to implement on farm. However, given the previous consumer reaction to the discovery of DCD residues in milk, there may still be reservations about its re-introduction even if the international standards are modified. Other NIs, such as 3,4-dimethylpyrazole phosphate, are still available for use. However, the contaminant scare with DCD highlights the need to find solutions to reducing GHG emissions without the risk of adverse effects on food production.

## 2.8 Interactions of greenhouse gas mitigation options

Interactions among animal, environment, management, production, and mitigation practices are inevitable and therefore, evaluation in controlled experimental conditions often result in unexpected outcomes when applied to the whole farm (Hristov *et al.*, 2013b). The effect of combining several mitigation strategies may not be additive. Knapp *et al.* (2014) combined four individual animal and feeding-management mitigation strategies, which individually could reduce enteric $CH_4$ (g $CH_4$/kg ECM) by between 5 and 18%. The theoretical sum of each mitigation strategy was over 50% reduction. However, the additive maximum reduction was lower, at only 30%.

It is also important that the reduction in one source of emission does not increase another source of emission, frequently termed ‘pollution swapping’. With a mitigation strategy such as NIs to reduce soil $N_2O$ emissions, the N is retained in a form where $NH_4$ accumulation can increase, consequently increasing the potential for $NH_3$ volatilisation and $NO_3$ leaching. While both gases are not a direct GHG, they act as a secondary source of $N_2O$ if soil conditions remain conducive for N loss, and thus the potential for pollution swapping. Reducing rumen degradable CP concentration in the diet, to reduce $N_2O$ emissions, needs to be achieved without a high fibre carbohydrate replacement (*e.g.* maize silage), as this has the potential to increase enteric $CH_4$ emissions (Dijkstra *et al.*, 2011). If a mitigation strategy increases plant N-use efficiency, resulting in greater pasture production, this must not facilitate an increases in stocking rate as GHG emissions per unit of land will increase if no other changes are made to the farm system (de Klein and Eckard, 2008). These counteracting effects need careful evaluation in the context of the whole farm system so that animal and pasture productivities are maximised without compromising farm profitability (Hristov *et al.*, 2013b; Rawnsley *et al.*, 2018).

### 2.9 Summary

The objectives of this thesis were to estimate the GHG emissions of the Australian dairy industry, reviewing the variability that occurs across the country and the effect of the evolution of methodologies and EFs to estimate GHG emissions. Benchmarking of farm GHG emissions allows for a reference point to be established. It determines where the farm's emissions currently are and provides a guide to where the emissions could decrease to. When comparing the emissions of farms, it is important that methodology, allocation, systems boundary and FU are taking into consideration as these can all have an influence on the EI of milk production. Benchmarking can also identify areas of inefficiency in the system and thus highlight mitigation options that could be implemented to reduce the EI of milk production. Mitigation options need to deliver win:win outcomes and must be considered in the system and global context to ensure implementation does not inadvertently create perverse outcomes.

While much of the literature review focussed on mitigation options to reducing the EI of milk production, the Paris Agreement stipulated anthropogenic emissions by sources will need to be balanced with sinks in the second half of the 21$^{st}$ Century to remain under a 2°C threshold temperature increase (UNFCCC, 2015). The current focus of the Australian dairy industry is on reducing the EI of milk production, which is vitally important. This is a tangible metric to compare within and across farms, allows the industry to grow by increasing the efficiency of production while reducing GHG emissions relative to business as usual. In addition to this, over the next 30 years, the industry will need to change its focus towards reducing net emissions by implementing best-practice mitigation and then offset the unavoidable, residual emissions through C sequestration to achieve zero net GHG emissions by 2050.

# CHAPTER 3 A WHOLE FARM SYSTEMS ANALYSIS OF GREENHOUSE GAS EMISSIONS OF 60 TASMANIAN DAIRY FARMS

Animal Feed Science and Technology 166–167 (2011) 653–662

Contents lists available at ScienceDirect

Animal Feed Science and Technology

journal homepage: www.elsevier.com/locate/anifeedsci

## A whole farm systems analysis of greenhouse gas emissions of 60 Tasmanian dairy farms

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ARTICLE INFO

*Keywords:*
Australia
Carbon dioxide
DGAS
Methane
Nitrous oxide
Pre-farm emissions

ABSTRACT

The Australian dairy industry contributes ~1.6% of the nation's greenhouse gas (GHG) emissions, emitting an estimated 8.9 million tonnes of $CO_2$ equivalents (t $CO_2$e)/annum (DCC, 2008). This study examined GHG emissions of 60 Tasmanian dairy farms using the Dairy Greenhouse gas Abatement Strategies (DGAS) calculator, which incorporates International Panel on Climate Change (IPCC) and Australian inventory methodologies, algorithms and emission factors. Sources of GHG emissions including pre-farm embedded emissions associated with key farm inputs (*i.e.*, grains/concentrates, forages and fertilizers) and on-farm emissions from $CO_2$, $CH_4$ and $N_2O$ are estimated by DGAS software. A detailed description of GHG calculations and functionality of DGAS software are provided. Total farm GHG emissions of 60 Tasmanian dairy farms, as estimated with DGAS, ranged between 704 and 5839 t $CO_2$e/annum, with a mean of 2811 t $CO_2$e/annum. Linear regression analyses showed that 0.93 of the difference in total farm GHG emission was explained by milk production. The estimated mean GHG emission intensity of milk of production was 1.04 kg $CO_2$e/kg fat and protein corrected milk (FPCM; ranged between 0.83 and 1.39 t $CO_2$e/t FPCM) with a standard deviation of 0.13. Stepwise multiple linear regression analysis showed that feed conversion efficiency (kg FPCM/kg dry matter (DM) intake) and N based fertilizer application rate explained 0.60 of the difference in the GHG emissions due to milk production from these pastoral based dairy systems. Estimated per cow and per hectare emission intensity was $6.9 \pm 1.46$ t $CO_2$e/cow and $12.6 \pm 4.37$ t $CO_2$e/ha, respectively. Stepwise multiple linear regression analysis showed that DM intake per cow (t DM intake/cow/lactation) explained 0.86 of the variability in per cow GHG emissions intensity, while milk production/hectare (t FPCM/ha) explained 0.92 of the variability in per hectare GHG emission intensity. Given the influence that feed conversion efficiency and/or N based fertilizer application rates had on all GHG emissions intensities, it is clear that these factors should be key target areas to lower the intensity of emissions associated with dairying in Tasmania.

*This paper is part of the special issue entitled: Greenhouse Gases in Animal Agriculture – Finding a Balance between Food and Emissions, Guest Edited by T.A. McAllister, Section Guest Editors: K.A. Beauchemin, X. Hao, S. McGinn and Editor for Animal Feed Science and Technology, P.H. Robinson.*



*Abbreviations:* $CO_2$e, $CO_2$ equivalent; CP, crude protein; DGAS, Dairy GHG Abatement Strategies; DM, dry matter; DMD, DM digestibility; EF, emission factor; FCE, feed conversion efficiency; FPCM, fat and protein corrected milk; GHG, greenhouse gas; IPCC, Intergovernmental Panel on Climate Change; SMLR, stepwise multiple linear regression.

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doi:10.1016/j.anifeedsci.2011.04.046

### 3.1 Introduction

Warming of the climate system is unequivocal, at least in the minds of most persons, as is evident from recent observations of increases in global average air and ocean temperatures, widespread melting of snow and ice and rising global sea levels (IPCC 2007). Most of the observed increase in global temperatures is likely due to the observed increase in anthropogenic greenhouse gas (GHG) emissions (IPCC 2007). It is estimated that global GHG emissions in 2005 totalled 40,950 million tonnes of $CO_2$ equivalents (t $CO_2$e), with Australia responsible for 553 t $CO_2$e, ~1.5% of world emissions (Climate Analysis Indicators Tool 2009).

The stationary energy sector was the largest source of GHG emissions in Australia in 2008, accounting for nearly half of this total, with agriculture the second largest contributor, accounting for ~16% of the nations' GHG emissions (DCC 2008). The livestock industries of dairy, beef and sheep farming contributed ~10, 47 and 19% of these agricultural GHG emissions, respectively (DCC 2008). The major source of GHG emissions from these livestock industries was $CH_4$ from enteric fermentation. Nitrous oxide ($N_2O$) emissions were also generated from N based fertilizers, animal deposition and manure management. In addition, there were indirect $N_2O$ emissions associated with losses to the environment through atmospheric volatilisation and runoff/leaching of N based fertilizers and animal waste. The Dairy Greenhouse gas Abatement Strategies (DGAS) calculator was developed to model $CO_2$, $CH_4$ and $N_2O$ associated with dairying in Australia, using Intergovernmental Panel on Climate Change (IPCC) and Australian national inventory methodologies, algorithms and emission factors (DCCEE 2009). While analysis of GHG emissions of dairy farm systems has been undertaken for the dairy industry of many countries, including Ireland (Casey and Holden 2005a), Sweden (Cederberg and Flysjö 2004) and New Zealand (Basset-Mens *et al.* 2005), there has been a paucity of studies undertaken under Australian conditions to estimate the GHG emissions among dairy farms.

Dairying is a well-established industry in Australia. While the bulk of milk production occurs along the coastal areas of the south-east corner of the country (i.e., 66% from Victoria, South Australia, New South Wales and Tasmania), the industry is also located in sub-tropical coastal northern New South Wales and Queensland, the south west of Western Australia and adjacent to major river systems in inland New

South Wales and Victoria. In 2008/2009, the dairy industry produced ~9.0 billion litres of milk from 1.7 million cows and 7800 farms (Dairy Australia 2009).

The current study determined GHG emissions, as estimated by the DGAS calculator, of 60 Tasmanian dairy farms. These farms were pasture based grazing systems with varying levels of milk production, grain feeding, N fertilizer application rates and reliance on irrigation water for pasture and crop production. This study also examined relationships between GHG emission intensity and some key farm variables.

## 3.2 Materials and methods

### *3.2.1. Farm system boundary, global warming potentials and data collection*

The farm system boundary for this study was defined by the GHG emissions associated with milk production up to the point of transportation from the farm, but included embedded pre-farm emissions. A global warming potential of 1, 21 and 310, respectively, was used to convert $CO_2$, $CH_4$ and $N_2O$ emissions into $CO_2e$ emissions (DCCEE 2009), as these are the current global warming potentials for the Australian inventory. The farm and herd farm physical and key farm input data from 60 Tasmanian dairy farms (~12% of the state industry) for the 2006/2007 milking year were collected during one-on-one farmer interviews. The mean and range of farm and herd input data for the 60 farms are shown in Tables 3.1 to 3.3. Farms were located in the north-east and north-west of the state, and represented the diversity of the industry in terms of milk production per cow, milking herd size, level of grain feeding and N based fertilizer usage. All areas used for dairy related activities, including the milking platform and run off areas for raising replacement stock and growing pastures and crops for forage conservation were included in total farm area. Milk production was reported as fat and protein corrected milk (FPCM), calculated as: FPCM (kg) = raw milk (kg; calculated by multiplying liters by 1.03 (Sevenster and de Jong 2008) × (0.337 + (0.116×fat content (g/100 g milk)) + (0.06×protein content (g/100 g milk)) (FAO 2010).

**Table 3.1** Key farm input data required in the Dairy Greenhouse gas Abatement Strategies calculator to estimate greenhouse gas emission (t $CO_2$e) and the mean (minimum and maximum in parenthesis) values for each of these key inputs for the 60 Tasmanian dairy farms.

| Key farm input data | Mean (min and max) |
|---|---|
| Farm area- total (ha) | 237 (57-576) |
| Farm area- irrigated (ha) | 72 (0-280) |
| Farm area- non-irrigated (ha) | 165 (0-576) |
| Milking platform area (ha) | 174 (57-375) |
| Electricity (000's kWh/yr) | 202 (17-608) |
| Diesel (000's L/annum) | 10.8 (0-43.1) |
| N fertilizer (000's kg N/yr) | 34.7 (1.5-138.9) |
| P fertilizer (000's kg P/yr) | 8.1 (0.5-33.3) |
| K fertilizer (000's kg K/yr) | 13.8 (0-99.4) |
| S fertilizer (000's kg S/yr) | 7.4 (0.7-27.8) |
| Purchased concentrates (t DM/yr) | 583 (0-1,560) |
| Purchased forage (t DM/yr) | 195 (0-1,029) |
| Purchased other feeds (t DM/yr) | 56 (0-722) |

$CO_2$e, carbon dioxide equivalent; DM, dry matter

**Table 3.2** Key herd input data required in the Dairy Greenhouse gas Abatement Strategies calculator to estimate greenhouse gas emission (t $CO_2e$) and the mean (minimum and maximum in parenthesis) values for each of these key inputs for the 60 Tasmanian dairy farms.

| Key herd input data | Mean (min and max) |
|---|---|
| Milking herd size (number of cows)[a] | 410 (147-870) |
| Milking herd average liveweight (kg) | 526 (420-650) |
| Rising 1 yr old replacement herd size | 118 (28-320) |
| Rising 2 yr old replacement herd size | 108 (0-255) |
| Mature bulls herd size | 9 (0-28) |
| Stocking rate (cows/ha) | 2.4 (1.1-4.1) |
| Total DMI (t DM/cow lactation$^{-1}$) [b] | 5.9 (3.9-7.6) |
| Concentrates (kg/cow yr$^{-1}$) | 1,452 (0- 2,920) |
| Pasture utilisation (t DM/ha)[c] | 9.3 (3.7-15.9) |
| Dietary dry matter digestibility (g/kg DM) | 699 (676-716) |
| Dietary crude protein (g/kg DM) | 191 (175-197) |
| Feed conversion efficiency (kg FPCM/kg DMI) [b] | 1.1 (0.8-1.3) |
| Percentage of grain in the milking herd diet | 23.3 (0-38.8) |

[a] Cows milked for more than 2 months and contributing to annual milk production.

[b] Total dry matter intake from home-grown pasture, conserved pasture, purchased forage, purchased grain/concentrates and purchased other feed sources (t DM intake/cow/lactation), as calculated using the Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009).

[c] Pasture utilisation (t DM consumed/ha), as estimated using the Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009).

$CO_2e$, carbon dioxide equivalent; DM, dry matter; DMI, dry matter intake; FPCM, fat and protein corrected milk

**Table 3.3** Key milk production input data required in the Dairy Greenhouse gas Abatement Strategies calculator to estimate greenhouse gas emission (t $CO_2$e) and the mean (minimum and maximum in parenthesis) values for each of these key inputs for the 60 Tasmanian dairy farms.

| Key milk production input data | Mean (min and max) |
|---|---|
| Milk production (000's kg FPCM/yr) | 2,734 (521-5,753) |
| Annual mean butterfat (g/100 g milk) | 4.2 (3.3-5.5) |
| Annual mean protein (g/100 g milk) | 3.4 (3.2-3.8) |
| Milk production (kg FPCM/cow yr $^{-1}$) | 6,775 (3,304-9,642) |
| Milk production (kg FPCM/ha yr $^{-1}$) | 12,332 (3,579-25,984) |

$CO_2$e, carbon dioxide equivalent; FPCM, fat and protein corrected milk

### *3.2.2. Pre-farm embedded emissions*

Simapro life cycle assessment software (Simapro 2006) was used to determine the $CO_2$e emissions associated with production of key farm imports and the associated emission factor (EF) for each is shown in Table 3.4. As each farm applied varying blends of fertilizer, each was converted into kg of N, P, K and S and then converted into the equivalent amount of urea (0.46 N), single superphosphate (0.09 P and 0.11 S) and potassium chloride (0.50 K). The amount of each feed type and fertilizer was multiplied by their corresponding EF, with results as GHG emissions (kg $CO_2$e) from fertilizer, grain/concentrates and other feed sources.

**Table 3.4** Greenhouse gas emissions factors for the production of concentrates and grains, hay and silage and fertilizer as calculated using Simapro software (Simapro 2006).

| **Key farm input** | **Emission factor (kg $CO_2$e/kg product)** |
|---|---|
| Grain/concentrates | 0.30 |
| Pasture hay and silage | 0.25 |
| Cereal/Maize silage | 0.25 |
| Lucerne hay | 0.20 |
| Urea | 0.89 |
| Single superphosphate | 0.23 |
| Potassium chloride | 0.13 |

$CO_2$e, carbon dioxide equivalent

### *3.2.3. Calculating on-farm $CO_2$ emissions*

In this study, on-farm $CO_2$ emissions were defined as those associated with electricity and diesel fuel consumption. Australian electricity is generated by a range of sources (e.g., brown and black coal, natural gas, hydro, and wind). However, as most of the country (including Tasmania) is connected to a national grid, it is difficult to know where or how the electricity is being generated. We selected the option with the highest EF (i.e., brown coal from Victoria) as the source of electricity, equivalent to 1.4 t $CO_2$e emitted for each 1000 kWh of electricity consumed (DCCEE 2009). The extraction/refinement and consumption of diesel fuel by farm vehicles, machinery and irrigation pumps was equivalent to 3.4 t $CO_2$e emitted for every 1000 L of fuel consumed (DCCEE 2009).

### *3.2.4. Calculating on-farm $CH_4$ emissions*

Methane is emitted on-farm from two main sources, being enteric fermentation and manure management. Enteric $CH_4$ was estimated for four stock classes (*i.e.,* milking herd, growing one year olds, growing two year olds, mature bulls), using data for each stock class liveweight, liveweight gain, milk production and mean annual diet dry matter (DM) digestibility (DMD; g/kg DM intake). Enteric fermentation was calculated in DGAS from a series of methodologies, algorithms and emission factors in the Australian National Greenhouse Accounts National Inventory Report (DCCEE 2009), based on research by Brouwer (1965), Blaxter and Clapperton (1965), Minson and McDonald (1987) and the Australian Standing Committee on Agriculture (1990).

Dry matter digestibility (g/kg DM) and crude protein (CP; g/kg DM) estimates were assigned to each feed source used, based on extensive results from the FeedTest® laboratory in Australia, as published in the Pasture Consumption and Feed Conversion Efficiency Calculator manual (Heard and Wales 2009; Table 3.5). These estimates where used to determine mean annual DMD (g/kg DM) and CP (g/kg DM) of the diet for the milking herd. For all farms, the replacement herd and mature bull herd was assumed to have a diet with a DMD of 650 g/kg DM and CP of 180 g/kg DM. The Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009) was also used to determine annual pasture utilization for the milking platform (t DM/ha) and annual DM intake (t DM intake/cow/lactation) for the milking herd, taking into consideration a feed out wastage for grains/concentrates, forage and other feed sources.

Methane from manure management was calculated in DGAS using a series of methodologies, algorithms and emission factors (DCCEE 2009), based on research by Williams (1993) and IPCC (1997) guidelines. In addition, an integrated $CH_4$ conversion factor was required, based on proportioning of animal waste to varying manure management regimes. For Tasmania, the manure management regime allocated 92% of waste voided onto pastures directly, 6% stored in a lagoon system, 1.5% spread on pastures daily and 0.5% stored as a liquid/slurry and applied later (DCCEE 2009).

**Table 3.5** Dry matter digestibility and crude protein (g/kg dry matter) values used for each feed source fed to the milking herd for each of the 60 Tasmanian dairy farms.

| Feed source | Dry matter digestibility (g/kg dry matter) | Crude protein (g/kg dry matter) |
|---|---|---|
| Home grown consumed pasture | 700 | 200 |
| Home grown conserved forage | 650 | 180 |
| Purchased forage | 650 | 180 |
| Grain | 800 | 120-190[a] |
| Other feed source | 600-750[b] | 180-240[b] |

[a] 22 farms fed grain with a crude protein of 120 g/kg dry matter while 38 farms fed a 70:15:15 grain/lupins/canola meal blend with a crude protein of 190 g/kg dry matter.

[b] Range of other feeds used so dry matter digestibility and crude protein based on each individual farm inputs.

### *3.2.5. Calculating on-farm $N_2O$ emissions*

Four sources of $N_2O$ emissions were estimated which were those associated with manure management, N based fertilizers, deposition of animal waste directly onto pastures during grazing and indirect $N_2O$ emissions associated with the potential for N based fertilizers, and animal waste to be lost to the environment through leaching/runoff and volatilisation. The manure management regime allocation fractions for Tasmania, as described earlier (Section 2.2.4), were also used to calculate $N_2O$ emissions associated with animal waste. Nitrogen based fertilizer $N_2O$ emissions were calculated based on emission factors using research by Galbally *et al.* (2005), and the application rates of N based fertilizer. Nitrous oxide emissions associated with feces and urine excretion were calculated, using methodologies, algorithms and emission factors that reflect Australian conditions (DCCEE 2009), based on research by the Australian Standing Committee on Agriculture (1990) and Freer *et al.* (1997). The proportion of N based fertilizers and animal waste that is available for direct and indirect $N_2O$ emissions and their corresponding EF is shown in Table 3.6.

**Table 3.6** Proportion of N based fertilizers and animal waste that is available for direct and indirect $N_2O$ emissions and their corresponding emission factor.

| | Source | Proportion available for loss to the environment | Emission factor |
|---|---|---|---|
| Direct $N_2O$ | N based fertilizer- (irrigated pastures and crops) | 1.0 | 0.4% |
| | N based fertilizer- (non-irrigated pastures and crops) | 1.0 | 0.3% |
| | Urine | 1.0 | 0.4% |
| | Faeces | 1.0 | 0.5% |
| | Stored manures | 1.0 | 1.8% |
| Indirect $N_2O$ - leached/runoff | N based fertilizers | 0.3 | 1.25% |
| | Animal waste | 0.3 | 1.25% |
| Indirect $N_2O$ - atmospheric deposition | N based fertilizers | 0.1 | 1.0% |
| | Animal waste | 0.07 to 0.4[a] | 1.0% |

[a] 0.07 for daily spread, 0.2 for voided directly onto pastures, 0.35 for stored in lagoons and spread later and 0.40 for liquid/slurry.

### *3.2.6. Dairy Greenhouse gas Abatement Strategies calculator*

The DGAS calculator (Advisor version 1) was constructed as Microsoft Excel and incorporates forms for ease of use. The calculator has, at its core, 5 user forms and 13 worksheets. The first two forms are for farm and herd data entry for the baseline farm system. The farm data includes farm size, including proportion of farm area that is used to grow pastures and crops and proportion of farm area that is irrigated, location, annual rainfall, area of tree plantings, manure management system, electricity, diesel, fertilizer usage and purchased feed inputs. Herd data includes milk production and 5 livestock classes including animal numbers, liveweight, liveweight gain and dietary composition. The diet for the milking herd allows for seasonal variation in dietary composition while diets of other livestock classes were fixed on an average annualized basis.

The third user form displays results of the baseline farm system, both as graphics and text. One functionality of DGAS is the opportunity to compare a baseline farm system with a hypothetical strategy farm system to ascertain impacts that GHG mitigation strategies have on farm GHG emissions. The last two forms are structured in a similar format to the first two, but allow for alterations to farm and/or herd data to assess implications of mitigation strategies on farm GHG emissions. The calculator presents baseline and strategy farm results to assess impacts that adopting the mitigation strategy will have on both total farm and milk intensity GHG emissions. The 13 worksheets incorporate the methodologies, algorithms and emission factors to calculate $CO_2e$ emissions associated with the embedded pre-farm inputs, and on-farm $CO_2$, $CH_4$ and $N_2O$ emissions. These worksheets can be altered, if required, to reflect changes to the methodologies, algorithms and/or emission factors.

*3.2.7. Statistical analysis*

Statistical Program for the Social Sciences Statistics (SPSS 2008) was used to regress total farm GHG emissions against milk production, cow numbers and total farm area. A stepwise multiple linear regression (SMLR) analysis between three measures of GHG emissions intensity and individual key farm variables used the statistical functions of SPSS Statistics. The three functional units of emissions intensity used were: emissions/kg of milk production (kg $CO_2e$/kg FPCM), emissions/milking cow (t $CO_2e$/cow) and emissions/unit of land (t $CO_2e$/ha). Key farm variables used in the SMLR analysis were milk production/cow (t FPCM/cow), milk production/ha (t FPCM/ha), stocking rate (number of cows/ha of milking platform), pasture utilisation (t DM consumed/ha), total feed intake (t DM intake/cow/lactation), feed conversion efficiency (FCE; kg FPCM/kg DM intake), proportion of grain in the milking herd diet and N fertilizer application rate (kg N fertilizer/ha).

Where the coefficient of determination of a linear regression is discussed, the result is reported as $r^2$. Where the coefficient of determination of a SMLR is discussed, the result is reported as $R^2$.

### 3.3 Results

The mean ± SD of farm GHG emission, as estimated by the DGAS calculator, was 2811 ± 1264 t $CO_2$e/annum. A positive linear relationship (Figure 3.1) existed between total farm GHG emissions and milk production (Eq. (1)), herd size (Eq (2)), and farm area (Eq (3)) as:

(1) Total GHG emissions (t $CO_2$e/annum) = fat protein corrected milk production (t FPCM) × 0.96 + 42.90; $r^2 = 0.93$ ($P < 0.001$)

(2) Total GHG emissions (t $CO_2$e/annum) = milking herd size (number of cows) × 5.94 + 373.75; $r^2 = 0.75$ ($P < 0.001$)

(3) Total GHG emissions (t $CO_2$e/annum) = total farm area (ha) × 7.68 + 993.95; $r^2 = 0.41$ ($P < 0.001$)

Estimated GHG emission intensity of milk production was 1.04 ± 0.13 kg $CO_2$e/kg FPCM, estimated GHG emissions intensity/cow was 6.9 ± 1.46 t $CO_2$e/cow, and estimated GHG emissions intensity/hectare was 12.6 ± 4.37 t $CO_2$e/ha.

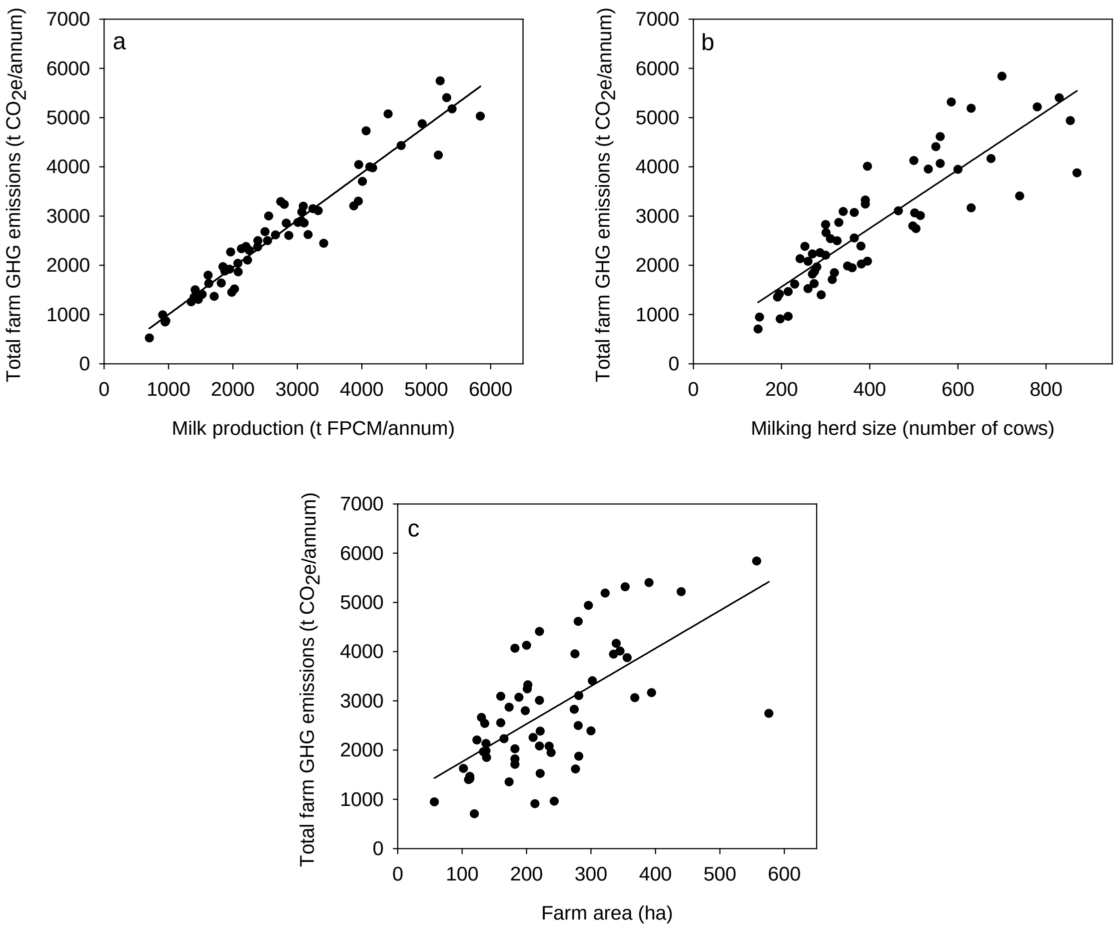


**Figure 3.1** Linear relationship between farm greenhouse gas (GHG) emissions (t $CO_2$e/annum), as estimated with the DGAS calculator, and milk production (a, total GHG emissions (t $CO_2$e/annum) = 0.96 x FPCM (t/annum) + 42.90; $r^2$=0.93), milking herd size (b, total GHG emissions (t $CO_2$e/annum) = 5.94 x milking herd size (number of cows) + 373.75; $r^2$=0.75) and farm area (c, total GHG emissions (t $CO_2$e/annum) = 7.68 x total farm area (ha) + 993.95; $r^2$=0.41).

The contribution of the various emission sources, as a proportion of total farm GHG emissions, is shown in Table 3.7. Enteric $CH_4$ was the biggest source of total farm GHG emissions. The next two largest sources were on-farm $CO_2$ from electricity and diesel consumption and indirect $N_2O$ emissions from N based fertilizers and animal waste.

**Table 3.7** Mean and range of individual greenhouse gas emissions sources, as a proportion of total farm greenhouse gas emissions for the 60 Tasmanian dairy farms as estimated by the DGAS calculator.

| Greenhouse gas emission source | Mean | Range |
|---|---|---|
| Enteric $CH_4$ | 0.551 | 0.455 – 0.669 |
| $CO_2$ from electricity and diesel | 0.114 | 0.031 – 0.220 |
| Indirect $N_2O$ from N-based fertilizers and animal waste | 0.109 | 0.088 – 0.143 |
| Direct $N_2O$ from animal waste[a] | 0.074 | 0.060 – 0.099 |
| $CO_2$ from purchased grain/concentrates | 0.061 | 0.000 – 0.098 |
| $CO_2$ from purchased fertilizers (N and non-N based) | 0.036 | 0.002 – 0.076 |
| Direct $N_2O$ from N-based fertilizers | 0.025 | 0.001 – 0.056 |
| $CO_2e$ from purchased forage | 0.018 | 0.000 – 0.068 |
| $CH_4$ from manure management | 0.013 | 0.011 – 0.016 |

[a] includes $N_2O$ emissions from manure management of stored manures (mean of 0.1%)

The SMLR analysis showed that FCE (kg FPCM/kg DM intake) alone explained 0.55 of the difference in the emission intensity of milk production (kg $CO_2e$/kg FPCM) among farms (Table 3.8). Addition of N based fertilizer application rates to the model accounted for an additional 0.05 of the difference in the milk intensity among farms (Table 3.8). The model that most accurately predicted milk intensity GHG emissions was:

(kg $CO_2e$/kg FPCM) = 2.03 + (-0.91 × FCE (kg FPCM/kg DM intake)) + (2.82E-04× N based fertilizer application rate (kg N/ha)).

Increases in FCE decreased GHG emission intensity of milk production while increases in N based fertilizer application rates increased the GHG emission intensity of milk production (Table 3.8).

**Table 3.8** Models of SMLR of the greenhouse gas emissions intensity expressed as milk intensity (kg $CO_2$e/kg FPCM), cow intensity (t $CO_2$e/cow) and area intensity (t $CO_2$e/ha), where *b* is the unstandardized coefficient, *SE b* is the standard error of *b, β* is the standardized coefficient and $R^2$ is the coefficient of determination.

| **Milk intensity (kg $CO_2$e/kg FPCM)** | | *B* | *SE b* | *β* | $R^2$ |
|---|---|---|---|---|---|
| Step 1 | Constant | 2.01 | 0.12 | | 0.55 |
| | Feed conversion efficiency (kg milk/kg DMI) | -0.85 | 0.10 | -0.74*** | |
| Step 2 | Constant | 2.03 | 0.11 | | 0.60 |
| | Feed conversion efficiency (kg milk/kg DMI) | -0.91 | 0.10 | -0.80*** | |
| | Nitrogen fertilizer (kg N/ha) | 2.82E-04 | 1.17E-04 | 0.21* | |
| **Cow intensity (t $CO_2$e/cow)** | | *B* | *SE b* | *B* | $R^2$ |
| Step 1 | Constant | -0.60 | 0.41 | | 0.86 |
| | Total feed intake (t DMI/cow/lactation) | 1.28 | 0.07 | 0.93*** | |
| Step 2 | Constant | -0.70 | 0.40 | | 0.87 |
| | Total feed intake (t DMI/cow/lactation) | 1.25 | 0.07 | 0.90*** | |
| | Nitrogen fertilizer (kg N/ha) | 1.89E-03 | 7.28E-04 | 0.13* | |
| Step 3 | Constant | -0.9 | 0.39 | | 0.89 |
| | Total feed intake (t DMI/cow/lactation) | 1.36 | 0.08 | 0.98*** | |
| | Nitrogen fertilizer (kg N/ha) | 3.03E-03 | 8.18E-04 | -0.20*** | |
| | Area production (t FPCM/ha) | -0.05 | 0.02 | -0.17* | |
| **Area intensity (t $CO_2$e/ha)** | | *B* | *SE b* | *B* | $R^2$ |
| Step 1 | Constant | 2.06 | 0.45 | | 0.92 |
| | Area production (t FPCM/ha) | 0.85 | 0.03 | 0.96*** | |
| Step 2 | Constant | 8.41 | 1.59 | | 0.94 |
| | Area production (t FPCM/ha) | 0.94 | 0.04 | 1.05*** | |
| | Feed conversion efficiency (kg milk/kg DMI) | -6.50 | 1.58 | -0.17*** | |
| Step 3 | Constant | 7.62 | 1.40 | | 0.95 |
| | Area production (t FPCM/ha) | 0.86 | 0.04 | 0.96*** | |
| | Feed conversion efficiency (kg milk/kg DMI) | -5.94 | 1.39 | -0.15*** | |
| | Nitrogen fertilizer (kg N/ha) | 6.76E-03 | 1.59E-03 | 0.15*** | |

Significant contributions to the model at * $P < 0.05$, ** $P < 0.01$; ***$P < 0.001$

The SMLR analysis showed that DM intake (t DM intake/cow/lactation) explained 0.86 of the differences in per cow GHG emissions intensity (t $CO_2$e/cow) among farms (Table 3.8). Addition of N based fertilizer application rates and milk production/ha improved prediction of per cow GHG emission intensity, although their addition could only account for an additional 0.03 of the difference (Table 3.8). Increases in DM intake and N based fertilizer application rates increased per cow GHG emission intensity while increases in milk production/hectare decreased per cow GHG emission intensity (Table 3.8).

The SMLR analysis showed that milk production/ha (t FPCM/ha) explained 0.92 of the difference in per hectare GHG emissions intensity (t $CO_2$e/ha) among farms (Table 3.8). Addition of FCE and N based fertilizer application rates improved prediction of per hectare GHG emission intensity, although their addition could only account for an additional 0.03 of the difference (Table 3.8). Consistent with the milk intensity emissions model, increases in FCE decreased per hectare GHG emission intensity, while increases in milk production per hectare and N based fertilizer application rates increased per hectare GHG emission intensity.

### 3.4 Discussion

Results show that milk production was an accurate way of predicting total farm GHG emissions since milk production accounted for 0.93 of the difference in estimated total farm GHG emissions. While this suggests that milk production alone is a suitable surrogate for estimating farm emissions from pasture based systems, the GHG emissions intensity of milk production varied between 0.83 and 1.39 kg $CO_2$e/kg milk. Only 0.60 of difference in milk GHG emissions intensity was explained using the key farm variables in the SMLR, and it was most strongly influenced by FCE and the amount of N based fertilizer applied. Given the strong influence that FCE and/or N based fertilizer application rates had on the variability of all three GHG emissions intensities, it is clear that these factors should be key target areas for lowering the extent of GHG emissions associated with dairying in Tasmania.

Improvements in FCE could be achieved through several mechanisms, including feeding options, such as improved herbage quality and improvements in animal performance through breeding (Clark *et al.* 2005). In a review of studies from the USA, New Zealand and Europe, Grainger and Goddard (2007) examined the differences in intakes and FCE between Holstein-Friesian and Jersey cows, fed both total mixed ration diets and predominantly pasture based diets. While Jersey DM intakes were always lower than those of Holstein-Friesians, FCE was similar or higher for Jersey cows compared to the Holstein-Friesians, for 9 of 11 studies. Improved FCE has been shown to influence $CH_4$ production. Clark *et al.* (2005) found that ruminants with a higher FCE produced 0.1 – 0.2 less $CH_4$ (g/kg DM intake) than those with a lower FCE.

Improvements in efficiency of use of N based fertilizers generally result in lower $N_2O$ emissions from soils. The rate, source and timing of N fertilizers have all been shown to influence $N_2O$ emissions (O'Hara *et al.* 2003). One example of reduced $N_2O$ emissions from N based fertilizer applications is use of nitrification inhibitors during times of the year when conditions are conducive to formation of $NO_3$ from $NH_3$ (*e.g.*, wet winters and springs). Research studies in New Zealand have shown that seasonal $N_2O$ emissions from N based fertilizers could be reduced by up to 80%, equivalent to 30 to 45% on an annual basis, with use of nitrification inhibitors (de Klein *et al.* 2001; Smith *et al.* 2008; Luo *et al.* 2010).

Although the proportion of concentrate in the diet was not a predictor in the SMLR analysis of emission intensity, it is well established that increasing the level of grain/concentrate in the diet reduces the proportion of dietary energy converted to $CH_4$ (Blaxter and Clapperton 1965). Lovett *et al.* (2006) found that increased grain feeding from 0.4 to 1.5 t DM/cow/lactation resulted in a decrease in milk GHG emissions by 0.11 kg $CO_2$e/kg milk. Similar results were reported by Johnson *et al.* (2002), who increased the proportion of concentrate in the diet from 40 to 370 g/kg DM, and observed a corresponding reduction in $CH_4$ production from 1.62 to 1.38 kg $CO_2$e/kg milk.

In our study, the proportion of grain/concentrate in the diet was 0 and 390 g/kg DM (0.0 and 2.9 t DM/cow/lactation), and there was a positive linear relationship (data not shown) between the proportion of grain in diet and FCE, with a 10% increase in grain in the diet equating to a 9% increase in FCE. As shown, an improvement in FCE resulted in a decline in the GHG emissions intensity of milk production. While feeding a high level of grain/cow can be profitable in some circumstances, detailed analysis of farming system performance in southern Australia has shown that farm profitability is more closely related to the amount of pasture consumed on a per hectare basis (Beca 2005; Savage and Lewis 2005; Chapman *et al.* 2008a).

Dairy farming in countries such as Ireland, New Zealand and Australia consumes a higher proportion of grazed pasture in the diet, which generally results in lower costs of production, compared with production costs of confined farming systems such as those in Canada, the USA and some European countries. Dillon *et al.* (2005) assessed relationships between milk production costs and the proportion of grazed pasture in the ration, and found that for every 10% increase in grazed pasture in the ration, milk production costs were reduced by 2.7 euro cents/liter.

While pasture consumption may be a good indicator of business performance (Beca 2005; Savage and Lewis 2005; Chapman *et al.* 2008a), our study found that it provides no indication of associated GHG emissions (data not shown). As such, feeding higher proportions of grain in the diet, as a management practice to improve per cow production, with the intention of reducing the GHG emission intensity of milk production, has the potential to reduce Australia's competitive advantage of producing milk at a lower cost of production compared to some of its international competitors.

Increasing the level of grain feeding corresponded with an increase in DM intake and per cow milk production (data not shown). Emission of $CH_4$ represents loss of dietary energy (Johnson *et al.* 1997; Lassey *et al.* 1997), and the algorithms and equations used to determine DM intake are based on milk production. Therefore, increased milk production and DM intake, due to increased grain feeding, directly increases enteric $CH_4$ production and per cow GHG emissions. Thus increasing the level of grain feeding may lead to increased stocking rates, thereby increasing enteric $CH_4$ emissions/unit land. Subsequently, although higher levels of grain feeding and corresponding high per cow production can reduce emissions due to milk production, these strategies will likely result in higher per farm, higher per cow and higher per unit of land GHG emissions.

Results from this study were congruent to studies from other countries. In a study comparing 10 dairy farms in Ireland, Casey and Holden (2005a) reported 0.92 – 1.51 kg $CO_2$e/kg milk, while Basset-Mens *et al.* (2005) found that the typical New Zealand dairy farm emitted 0.72 kg $CO_2$e/kg milk. Results from 23 conventional and organic farms in Sweden were 0.90 – 1.04 kg $CO_2$e/kg milk (Cederberg and Flysjö

2004), while results from Germany comparing 18 farms found that the GHG emissions were 1.0 – 1.3 kg $CO_2$e/kg milk (Haas *et al.* 2001). A more recent study by the Food and Agriculture Organization of the United Nations (FAO 2010), found that the global average of GHG emissions was 2.4 kg $CO_2$e/kg milk at the farm gate. However, there were substantial regional variations, ranging from a low of 1.0 – 1.5 kg $CO_2$e/kg milk for western industrialized regions (*e.g.,* USA, Eastern and Western Europe) to a high of 7.5 kg $CO_2$e/kg milk for the sun-Saharan region of Africa (FAO 2010). Oceania (dominated by the Australian and New Zealand dairy industries) was ~1.2 kg $CO_2$e/kg milk, reaffirming that results from our study were comparative to other international studies.

Comparing GHG emissions among countries is difficult and uncertain given the impact that different methodologies, emission factors and assumptions can have on the calculations. Some international studies allocate between 85 and 90% of total farm GHG emissions to milk production, with the balance allocated to meat production from cull cows, surplus heifers and bull calves (*e.g.,* Cederberg and Flysjö 2004; Basset-Mens *et al.* 2005). However, other studies (*e.g.,* Haas *et al.* 2001) and the current study have allocated all farm GHG emissions to the primary product milk. These differences in allocation of farm GHG emissions to milk and meat need to be considered when comparing results among studies.

There are also differences in methodologies and emission factors among countries. For example, in Australia the emission factor for direct $N_2O$ emissions from fertilizers was reduced from the IPCC based 1.25% emission factor (IPCC 2000b), to 0.4% for pastures and 0.3% for crops (DCCEE 2009). In New Zealand, an emission factor of 1.0% is used for direct $N_2O$ emissions from N based fertilizers (New Zealand Ministry for the Environment 2009). So, in effect, applying the same level of N based fertilizers in Australia, for example, would result in substantially lower direct $N_2O$ fertilizer emissions than it would in New Zealand. While the comparison of results from farms from the same country can be useful in identifying potential areas of abatement, diligence should be shown when comparing results using differing empirical methodologies.

While the empirical methodologies used in our study are accepted methods to account for farm GHG emissions, they may not be a precise assessment of actual on farm GHG emissions. Errors in data collection can influence the outcome of inventory assessments of GHG emissions, especially in areas that have been shown to influence GHG emissions intensity. While milk production figures can generally be relatively accurately collected, based on the volume of milk sold to milk processors, DM intake is less accurately predicted and based on numerous assumptions. The algorithms and emission factors can also be a source of error. For example, while based on the estimated $N_2O$ emissions under best management practices, allocation of a single emission factor for N fertilizer usage does not allow for the variations in soil types, rainfall patterns, or between rate, source and timing of applications.

### 3.5 Conclusion

Results show that GHG emissions of 60 Tasmanian dairy farms, as estimated using the DGAS calculator, could be accurately explained using a regression equation based on annual milk production. For each kilogram of fat and protein corrected milk produced, there was a corresponding total farm GHG emission of 0.96 kg $CO_2$e. Results from this study were similar to studies in other countries, thus illustrating that the pasture dominant farming systems in Tasmania were as GHG efficient as other pasture-based farming systems in New Zealand, Ireland and Europe.

### 3.6 Acknowledgements

This project was supported by funding from Meat & Livestock Australia, Dairy Australia, Australian Wool Innovation, the Tasmanian Institute of Agricultural Research and the Australian Government Department of Agriculture, Fisheries and Forestry under its Australia’s Farming Future Climate Change Research Program. The authors thank Dairy Australia for providing data for the 60 dairy farms and the contribution of data collection by the Dairy Business Centre and TIAR. The DGAS calculator is freely available and can be downloaded at www.dairyingfortomorrow.com.au/index.php?id=47.

# CHAPTER 4 WHOLE-FARM SYSTEMS ANALYSIS OF AUSTRALIAN DAIRY FARM GREENHOUSE GAS EMISSIONS

**CSIRO** PUBLISHING

*Animal Production Science*, 2012, **52**, 998–1011
http://dx.doi.org/10.1071/AN12061

## Whole-farm systems analysis of Australian dairy farm greenhouse gas emissions

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**Abstract.** The Australian dairy industry contributes ~1.6% of the nation's greenhouse gas (GHG) emissions, emitting an estimated 9.3 million tonnes of carbon dioxide equivalents ($CO_2e$) per annum. This study examined 41 contrasting Australian dairy farms for their GHG emissions using the Dairy Greenhouse Gas Abatement Strategies calculator, which incorporates Intergovernmental Panel on Climate Change and Australian inventory methodologies, algorithms and emission factors. Sources of GHG emissions included were pre-farm embedded emissions associated with key farm inputs (i.e. grains and concentrates, forages and fertilisers), $CO_2$ emissions from electricity and fuel consumption, methane emissions from enteric fermentation and animal waste management, and nitrous oxide emissions from animal waste management and nitrogen fertilisers. The estimated mean (±s.d.) GHG emissions intensity was $1.04 \pm 0.17$ kg $CO_2$ equivalents/kg of fat and protein-corrected milk (kg $CO_2e$/kg FPCM). Enteric methane emissions were found to be approximately half of total farm emissions. Linear regression analysis showed that 95% of the variation in total farm GHG emissions could be explained by annual milk production. While the results of this study suggest that milk production alone could be a suitable surrogate for estimating GHG emissions for national inventory purposes, the GHG emissions intensity of milk production, on an individual farm basis, was shown to vary by over 100% (0.76–1.68 kg $CO_2e$/kg FPCM). It is clear that using a single emissions factor, such as milk production alone, to estimate any given individual farm's GHG emissions, has the potential to either substantially under- or overestimate individual farms' GHG emissions.

Received 17 February 2012, accepted 15 May 2012, published online 2 August 2012

### 4.1 Introduction

The dairy industry is one of Australia's major rural industries, ranked third behind beef and wheat, producing ~ 9.0 billion litres of milk from 1.6 million cows on 7500 farms (Dairy Australia 2010a). South-eastern Australia's climate and natural resources are generally favourable to dairying, with ~ 66% of milk production coming from coastal regions of Victoria, South Australia, New South Wales and Tasmania (Dairy Australia 2010b). The industry is also located in subtropical coastal northern New South Wales and Queensland, the south west of Western Australia and adjacent to major river systems in inland southern New South Wales and northern Victoria. Over the last two decades, farm numbers have declined by ~ 40%, however, with increases in herd sizes and milk production per cow, Australia's annual milk production has increased from 6.2 billion litres in 1990 to a peak of 10.8 billion litres by 2000. In the last decade, Australian milk production has remained relatively stable at 9 to 10 billion litres (Dairy Australia 2010a).

The dairy industry is predominantly pasture-based, with ~ 70% of feed requirements coming from grazed pastures (Dairy Australia 2010a), although increases in farm intensification has largely been achieved through greater reliance on supplementary feeding and increase usage of nitrogen (N) fertiliser (Thorrold and Doyle 2007). Reliance on supplementary feed has allowed some farms to milk in excess of 1000 cows and there has been an increase in the establishment of feedlot dairies, particularly in traditional cropping regions (Dairy Australia 2010a). This intensification of the industry has also brought increasing focus on the environmental sustainability of dairying (Gourley 2004, Hart *et al.* 2004; Dougherty *et al.* 2008; Gourley *et al.* 2012a; 2012b) with greenhouse gas (GHG) emissions becoming another area of environmental importance (de Klein and Eckard 2008).

In 2009, Australia's agricultural sector accounted for ~ 15% of the nations' GHG emissions, the second largest contributor, behind stationary energy (DCCEE 2011a). The livestock industries of dairy, beef and sheep farming contribute ~ 10, 47 and 19% of these agricultural emissions, respectively (DCCEE 2011a). Although direct emissions from agricultural operations [e.g. methane ($CH_4$) emissions from cows or nitrous oxide ($N_2O$) emissions from animal waste and N fertiliser use] will not be subject to the price on carbon emissions in Australia (DCCEE 2011b), agriculture will have the option of providing emission offsets to other sectors (DCCEE 2010) through the Carbon Farming Initiative (CFI). The CFI is the proposed mechanism for assisting agricultural farmers and land managers to obtain carbon offset credits by sequestering carbon or by reducing/avoiding GHG emissions (DCCEE 2011c). If farmers and land managers choose to undertake a CFI approved project, rigorous methodologies will need to be adhered to (e.g. proof of the abatement being measurable and verifiable, permanent removal of GHG emissions, abatement is additional to 'business as usual' farm practices etc) before farmers and land managers will be allocated carbon offset credits. These credits can then be tradable so that other sectors of the economy can offset a portion of their GHG emissions (DCCEE 2011c). However, before agriculture can begin to reduce their GHG emissions to provide offsets for other sectors, there is a critical need to evaluate farm emissions, determine the sources of these emissions and their corresponding contribution and quantify the key management factors influencing emissions across differing farming systems (FS).

There have been assessments of either real or simulated beef and sheep enterprises for their GHG emissions profile (e.g. Kopke *et al.* 2008; Biswas *et al.* 2010; Peters *et al.* 2010; Browne *et al.* 2011). However, to date there have been few assessments of dairy farm GHG emissions across the various dairying regions of Australia, operating under different levels of farm intensity and management practices. The Victorian Department of Primary Industry have assessed the GHG emissions of between 57 and 73 dairy farms from northern, south-eastern and south-western Victoria for the last 4 years (2006-07 to 2009-10; English 2007; English *et al.* 2008; Gilmour *et al.* 2009, 2010) using the National Greenhouse Gas Inventories (NGGI) methodology (DCCEE 2009). However, they did not include any of the pre-farm embedded emissions associated with key farm inputs and so could not be considered as a whole farm systems approach. Beldman and Daatselaar (2010) followed NGGI methodology and included pre-farm embedded emissions but only assessed three dairy farms (Western Australia, northern and south-eastern Victoria). Christie *et al.* (2011) assessed 60 dairy farms' GHG emissions using the NGGI methodology, with the inclusion of pre-farm embedded emissions. However, all farms were located in a single region (Tasmania) and therefore exploring regional differences and their influence on GHG emissions was not possible. Therefore, to date, there has been limited assessment of dairy farm GHG emissions following the NGGI methodology, including pre-farm embedded emissions, across the various dairying regions of Australia, operating under different levels of farm intensity and management practices.

The aim of the current study was to estimate total farm GHG emissions of 41 Australian dairy farms from diverse geographical locations, varying herd and farm sizes, levels of milk production per cow and per hectare, and reliance on irrigation and supplementary feeding. This study also aimed to ascertain any regional differences in terms of three metrics; GHG emissions intensity per unit of milk produced, per cow and per hectare. In addition, this study examined the influence of key farm variables on these three abovementioned metrics.

## 4.2 Materials and methods

### *4.2.1. Farm selection and dataset*

The current study was designed to estimate the GHG emissions across the breadth of the Australian dairy industry and to enable a comparison of contrasting dairy systems. To achieve this, 41 Australian dairy farms were selected using a stratified-random process taking into consideration key criteria of (i) geographical location, (ii) litres of milk per grazed hectare, (iii) grazed hectares, and (iv) proportion of grazed hectares that were irrigated (Gourley *et al.* 2012b). Farms selected were representative of the local industry and varied in terms of milking herd size and farm size, level of milk production per cow, level of grain and other supplementary feeding and fertiliser inputs (Tables 4.1 to 4.3). This farm selection process resulted in a diversity of locations and FS to provide an industry-wide assessment of the current GHG emissions at a range of scales (e.g. range of milking herd sizes, farm areas, stocking rates, level of milk production per cow and level of supplementary feeding). Ten farms were located in south-eastern Victoria, nine farms in New South Wales, five farms in Western Australia, four farms in Queensland and Tasmania and three farms in South Australia, south-western Victoria and northern Victoria.

**Table 4.1** The mean (minimum and maximum in parenthesis) farm values required to estimate greenhouse gas emissions.

| Key farm input data | Mean (min and max) |
|---|---|
| Farm area- total (ha) | 338.6 (67.3 – 1045.6) |
| Farm area- milking platform (ha) | 191.7 (52 – 460) |
| Farm area- irrigated (ha) | 63.2 (0 – 329) |
| Farm area- non-irrigated (ha) | 128.6 (3 – 460) |
| Electricity (000's kWh/yr) | 145.8 (27.2 – 1023.1) |
| Diesel (000's L/annum) | 9.6 (6.2 – 25.4) |
| N fertiliser (000's kg N/yr) | 23.4 (0.0 – 154.3) |
| P fertiliser (000's kg P/yr) | 4.4 (0.0 – 25.1) |
| K fertiliser (000's kg K/yr) | 8.5 (0.0 – 64.4) |
| S fertiliser (000's kg S/yr) | 4.1 (0.0 – 26.0) |
| Purchased concentrates (t DM/yr) | 436.1 (19.9 – 2336.6) |
| Purchased forage (t DM/yr) | 233.4 (0.0 – 1788.7) |
| Purchased other feeds (t DM/yr) | 132.7 (0.0 – 2375.9) |

$CO_2e$, carbon dioxide equivalent; DM, dry matter

**Table 4.2** The mean (minimum and maximum in parenthesis) herd values required to estimate greenhouse gas emissions.

| Key herd input data | Mean (min and max) |
|---|---|
| Milking herd size (number of cows)[a] | 355 (62 – 1350) |
| Milking herd average liveweight (kg) | 534 (453 – 550) |
| Heifer herd size (number of rising 1 and 2 yr olds) | 72 (14 – 190) |
| Replacement rate (%) | 22.1 (3.9 – 36.1) |
| Mature bulls herd size | 7 (0 – 40) |
| Number of bulls per 100 milkers | 2.0 (0.0 – 6.6) |
| Stocking rate (cows/ha) | 2.0 90.6 – 4.4) |
| Pasture consumption (t DM/ha) [b] | 6.5 (0.1 – 14.1) |
| Concentrates (t DM/cow.lactation) | 1.3 (0.0 – 2.9) |
| Estimated total DMI (t DM/cow lactation$^{-1}$) [b] | 5.8 (3.6 – 7.8) |
| Dietary dry matter digestibility (%) | 74.5 (68.9 – 78.9) |
| Dietary crude protein (%) | 19.8 (14.4 – 24.4) |
| Feed conversion efficiency (litres of milk/kg DMI) | 1.04 (0.55 – 1.56) |
| Percentage of grain in the milking herd diet | 22.3 (0.0 – 57.4) |

[a] Cows milked for more than 2 months and contributing to annual milk production.

[b] Total dry matter intake from home-grown pasture, conserved pasture, purchased forage, purchased grain/concentrates and purchased other feed sources (t DM intake/cow/lactation), as calculated using the Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009).

DM, dry matter; DMI, dry matter intake

**Table 4.3** The mean (minimum and maximum in parenthesis) milk production values required to estimate greenhouse gas emissions.

| Key milk production input data | Mean (min and max) |
|---|---|
| Milk production (000's litres/year) | 2183 (373 – 11248) |
| Milk production (000's kg MS/year) | 160 (26 – 780) |
| Milk production (000's kg FPCM/year) | 2254 (374 – 11067) |
| Milk production (000's kg FPCM/cow.lactation) | 6.27 (3.25 – 9.87) |
| Milk production (000's kg FPCM/ha milking platform) | 12.67 (3.20 – 36.05) |
| Milk production (000's kg FPCM/ha total farm) | 7.62 (1.41 – 18.08) |
| Annual mean butterfat (g/100 g milk) | 4.10 (3.65 – 5.13) |
| Annual mean protein (g/100 g milk) | 3.31 (3.09 – 3.83) |

MS, milksolids; FPCM, fat and protein corrected milk

Farms were visited five times throughout the 12-month study period (February 2008 to February 2009) with visits identified as being $T_0$ (summer 2008) at the commencement and $T_1$ (autumn 2008), $T_2$ (winter 2008), $T_3$ (spring 2008) and $T_4$ (summer 2009) occurring at the 3rd-, 6th-, 9th- and 12th- month stage of the study period. To establish an inventory of supplementary feeds present on the farm during the study period, the amount of conserved forage, grain and other feeds present at visit $T_0$ and $T_4$ were determined. Any home-grown conserved feed or purchased supplementary feed present at $T_0$ and consumed within the study period was included in diet intake estimations. Any home-grown conserved feed or purchased supplementary feed present at $T_4$ was excluded from the diet calculations as it was not consumed within the study period. This resulted in a closed system where the feed inventory was reflective of the conserved and purchased feed consumed within the study period. All feed purchased during the 12 month study period was classified as an import for pre-farm embedded emissions estimations, irrespective of whether it was or was not consumed during the study period.

At each visit, stock numbers present on the milking platform (i.e. area where generally only the milkers and bulls are located but could also include some or all of the rising 1- and 2-year-old replacement stock and non-lactating mature cows) and any runoff/outblock or leased areas (i.e. area where the rising 1- and 2-year-old replacement stock and non-lactating mature cows are generally located in addition to areas where supplementary feeds are grown, harvested and transported to the milking area) were recorded. For farms with one or two calving periods per annum, the maximum milking herd size from the five visits was used as the milking herd size for GHG emissions estimations. For farms with year-round calving, the milking herd and non-lactating mature cow herd were added together to provide a seasonal milking herd size for each visit. The milking herd size for GHG emissions estimations for year-round calving herds was taken as the second highest figure recorded during the five seasonal farm visits. This eliminated a potential overestimation of the milking herd size for year-round calving herds.

The number of first-lactation cows was used as the herd size for the rising 1- and 2-year-old replacement stock numbers. Some farms retained bulls year round while others only had bulls present during the breeding season (i.e. 1 – 2 visits). An average bull herd size was calculated based on bulls being present year round. The liveweight for the milking herd for each farm was based on the breed of cattle; 450 kg for Jerseys, 550 kg for Holstein-Friesians and 500 kg for all other breeds and Holstein-Friesian crossbreds (Dairy Australia 2003). For any herds with two or more breeds, a mean herd liveweight was calculated taking into consideration the number of milkers from each breed. The liveweight of the rising 1- and 2-year-old replacement stock were assumed to be 35 and 75% of the milking herd live weight (Dairy Australia 2003), respectively, while the bulls were assumed to be 650kg, irrespective of breed. Live weight gain was set at 0.7 kg/day for the rising 1 and 2 year old replacement stock (Dairy Australia 2003) and at 0 kg/day for the bulls and mature cows (based on the assumption that any loss of condition post calving is gained in mid to late lactation and so over the 12-month study period, the net weight gain is zero).

Daily grazed pasture and supplementary feed DM intakes for the milking herd were provided by the farmer at each visit. A sample of each feed source (pastures and supplements) fed to the milking herd on the day of each visit was collected, prepared and analysed for various feed quality parameters by George Weston Technologies (Enfield, NSW, Australia). The key feed quality parameters obtained and used in this study were crude protein concentration (CP; g/100 g DM) and metabolisable energy

(ME; MJ ME/kg DM). Dry matter digestibility concentration (DMD; g/ 100 g DM) was calculated from the obtained ME values using the following equation:

Dry matter digestibility (g/ 100 g DM) = (ME + 1.037) / 0.1604 (Minson and McDonald 1987) (1)

The DM intake (kg DM/day), DMD (g/ 100 g DM) and CP (g/ 100 g DM) for each component of the diet (pasture and supplementary feeds) was entered into the Dairy Greenhouse Gas Abatement Strategies (DGAS) calculator to calculate a mean seasonal diet DMD (g/ 100 g DM) and CP (g/ 100 g DM) for the milking herd. The milking herd's diet DMD and CP (g/ 100 g DM) was calculated based on the dietary information collated during visits $T_1$ – $T_4$ and used throughout DGAS (as required) to estimate $CH_4$ and $N_2O$ emissions. Total DM intake (t DM/cow.lactation) and pasture consumption (t DM consumed/ha) were estimated using the Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009) and used in the stepwise multiple linear regression (SMLR) analysis. No diet information (quality or quantity) was collected for rising 1- and 2-year-old replacement stock or the bulls. This study assumed the mean annual DMD and CP was 70 and 18 g/100 g DM, respectively, for all non-milking stock classes.

Monthly milk volume, mean butterfat % and mean protein % was provided by the various milk companies supplied by the participating farms. Mean annual butterfat % and protein % was calculated by summing the quotient of monthly milk volume by its corresponding milk component and dividing by total annual volume. To compare milk production between farms, fat and protein corrected milk (FPCM) was used to correct milk volume to a standard of 4.0% fat and 3.3% protein. This is a standard used for comparing milk with different fat and protein contents. It is a means of evaluating milk production of different dairy breeds on a common basis (FAO 2010). The annual fat and protein correct milk (FPCM) was calculated as:

FPCM (kg) = raw milk (litres x 1.03) (Sevenster and de Jong 2008) x [0.337 + (0.116 x fat content) + (0.06 x protein content)] (FAO 2010) (2)

where FPCM is in kg, raw milk is in litres and fat content and protein content are g/100 g milk.

There was no direct assessment of electricity and diesel consumption. Electricity consumption for milk harvesting was estimated at 0.67 kWh/cow.day (adapted from Genesis Now 1997) while electricity for irrigation was estimated at 200 or 275 (kWh/ML) for flood and spray delivery, respectively (adapted from NSW Department of Primary Industries 2003). Diesel consumption (in litres; adapted from Christie *et al.* 2011) was estimated as:

$$\text{Diesel (l)} = 25.5 \times \text{milk production (t MS/farm)} + 5500 \tag{3}$$

### *4.2.2. Greenhouse gas emissions estimation*

The farm system boundary for this study was defined by the GHG emissions associated with milk production up to the point of transportation from the farm, including pre-farm embedded emissions. All areas used for dairy related activities, including the milking platform and runoff/outblock or leased areas for raising young stock and growing pastures and crops for forage conservation were included in the total farm area. The DGAS (version 1.3) calculator was used to estimate GHG emissions using a global warming potential of 1, 21 and 310 to convert $CO_2$, $CH_4$ and $N_2O$ emissions into $CO_2$ equivalent ($CO_2e$) emissions, respectively (DCCEE 2009). The DGAS calculator incorporates the Australian NGGI methodology (DCCEE 2009) to estimate on-farm emissions ($CH_4$, $N_2O$ and $CO_2$ from energy). In addition, the DGAS calculator also incorporates calculations of $CH_4$, $N_2O$ and $CO_2$ emitted in the production/manufacturing of key farm inputs (i.e. supplementary feeds and fertilisers). The NGGI methodology complies with rules that conform to international guidelines adopted by the United Nations Framework Convention on Climate Change (DCCEE 2009). The NGGI methodology also conforms to the protocol required for the Australian Government to report the nation's annual anthropogenic sources and sinks as part of its commitments under the Kyoto Protocol (DCCEE 2009). The NGGI methodology has also been widely used to estimate GHG emissions from the agricultural sector (e.g. Petersen *et al.* 2003; Flugge and Schilizzi 2005; Biswas *et al.* 2010; Peters *et al.* 2010; Browne *et al.* 2011; Eady *et al.* 2011) and therefore is the most currently accepted approach for estimating GHG emissions for Australian dairy farms. All equations and constants relating to the GHG emissions estimations in this study are from the NGGI methodology (DCCEE 2009) unless stated otherwise.

### *4.2.3. Pre-farm embedded emissions*

Simapro life cycle assessment software (Simapro 2006) was used to determine the $CO_2$e emissions associated with the production of key farm imports. The amount of N, phosphorus (P) and potassium (K) fertiliser applied during the study period was converted into equivalent amounts of urea (46% N), triple superphosphate (18% P) and potassium chloride (50% K) and multiplied by their corresponding emission factor of 0.89, 0.83 and 0.13 kg $CO_2$e/kg product, respectively. The amount of purchased grains/concentrates, hay and silage was multiplied by their corresponding emission factor. These emission factors were 0.20 kg $CO_2$e/kg DM for lucerne hay, 0.25 kg $CO_2$e/kg DM for pasture and cereal hay and silage, and 0.30 kg $CO_2$e/kg DM for grains/concentrates. All by-products such as canola meal, brewer's grain and molasses were assumed to have no carbon footprint as the carbon liability was assumed to lie with the primary process (i.e. cooking oil production, beer brewing and sugar refining for the abovementioned by-products). The pre-farm embedded emissions were presented in terms of GHG emissions (t $CO_2$e) from fertiliser, grain/concentrates and forage sources.

### *4.2.4. Calculating on-farm carbon dioxide emissions*

Australian electricity is generated by a range of sources (e.g. brown and black coal, natural gas, hydro, solar, wind). However, as most of the country is connected to a national grid, it is difficult to know where or how electricity is being generated for individual regions. We selected brown coal as the source of electricity for all farms, with an emission factor of 1.4 kg $CO_2$e/kWh, with an exception for Western Australia farms, where natural gas was selected with an emission factor of 0.5 kg $CO_2$e/kWh (DCCEE 2009). The extraction/refinement and consumption of diesel fuel by farm vehicles and machinery was equivalent to 3.4 kg $CO_2$e/L (DCCEE 2009). The GHG emissions associated with transportation of key farm inputs was not taken into consideration in this study due to this information not being gathered from farmers during farm visits.

### *4.2.5. Calculating on-farm methane emissions*

Methane is emitted on farm from two sources; enteric fermentation and animal waste. Enteric fermentation was estimated in DGAS from a series of methodologies, algorithms and emission factors from the NGGI report (DCCEE 2009), based on an approach developed by Blaxter and Clapperton (1965), incorporating research by Minson and McDonald (1987) and the Standing Committee on Agriculture (1990). Throughout the DCCEE (2009) methodology, the Australian dairy industry is divided into subcategories for the estimation of GHG emissions, with these subcategories reported as subscript letters in the equations. The subscript i represents the various states of Australia (i.e. Queensland, Victoria, Western Australia etc), the subscript j represents the dairy cattle stock class (i.e. milking cows, heifers < 1 year of age, bulls > 1 year of age etc) and the subscript k represents season (i.e. spring, summer etc).

To estimate enteric $CH_4$ production, daily DM intake ($I_{ijk}$; kg DM/head.day) is calculated as:

$$I_{ijk} \text{ (kg DM/head.day)} = (1.185 + 0.00454 \text{ x } W_{ijk} - 0.0000026 \text{ x } W_{ijk}^2 + 0.315 \text{ x } LWG_{ijk})^2 \text{ x MR} + MI_{ijk} \quad (4)$$

where $W_{ijk}$ = live weight (kg), $LWG_{ijk}$ = live weight gain (kg/day), MR = metabolic rate when producing milk (1.1 for mature cows and 1.0 for all other classes) and $MI_{ijk}$ = additional intake required for milk production (kg DM/head.day; as calculated in Eqn 5 below).

The additional intake required for milk production ($MI_{ijk}$; kg DM/head.day) is calculated as:

$$MI_{ijk} \text{ (kg DM/head.day)} = MP_{ijk} \text{ x NE / (k x q x 18.4)} \quad (5)$$

where $MP_{ijk}$ = milk production (kg/head.day), NE = 3.054 MJ (net energy/kg milk; SCA 1990), k = 0.60 (efficiency of use of metabolisable energy for milk production), q = metabolisability of the diet [where q = 0.00795 x $DMD_{ijk}$ (g/ 100 g DM) of the diet – 0.0014] and 18.4 = gross energy content of feed [MJ/kg DM; SCA 1990 (where this value is assumed value for all feeds (DCCEE 2009)].

The gross energy intake ($GEI_{ijk}$; MJ/head.day) is calculated as:

$$GEI_{ijk} \text{ (MJ/ head.day)} = I_{ijk} \text{ x } 18.4 \quad (6)$$

where $I_{ijk}$ = daily intake (kg DM/head.day as calculated in Eqn 4 above) and 18.4 = gross energy content of feed (MJ/kg DM; SCA 1990).

Intake relative to that required for maintenance for each stock class ($L_{ijk}$) is calculated as:

$$L_{ijk} = I_{ijk} / (1.185 \times 0.00454 \times W_{ijk} – 0.0000026 \times W_{ijk}^2 + 0.315 \times LWG_{ijk})^2 \quad (7)$$

where $W_{ijk}$ = live weight (kg) and $LWG_{ijk}$ is set to zero.

The percentage of gross energy intake ($GEI_{ijk}$) that is yielded as enteric $CH_4$ ($Y_{ijk}$) is given by Blaxter and Clapperton (1965) and calculated as:

$$Y_{ijk}\% = 1.3 + 0.112 \times DMD_{ijk} + L_{ijk} \times [2.37 – (0.050 \times DMD_{ijk})] \quad (8)$$

where $DMD_{ijk}$ = dry matter digestibility (g/ 100 g DM) of the diet and $L_{ijk}$ = intake relative to that required for maintenance (as calculated in Eqn 7 above).

The total daily production of enteric $CH_4$ ($M_{ijk}$ enteric $CH_4$; kg $CH_4$/head.day) is calculated as:

$$M_{ijk} \text{ enteric } CH_4 \text{ (kg } CH_4\text{/head.day)} = (Y_{ijk} / 100) \times (GEI_{ijk} / F) \quad (9)$$

where $Y_{ijk}\%$ = percentage of gross energy intake yielded as enteric $CH_4$ (as calculated in Eqn 8 above), $GEI_{ijk}$ = gross energy intake (MJ/head.day; as calculated in Eqn 6 above) and F = 55.22 (MJ/kg $CH_4$; Brouwer 1965).

From this, total enteric $CH_4$ production (Gg $CH_4$/annum) is calculated as:

$$\text{Total enteric } CH_4 \text{ production (Gg } CH_4\text{/annum)} = \sum (M_{ijk} \text{ enteric } CH_4 \times N_{ijk} \times 365) \times 10^{-6} \quad (10)$$

where $M_{ijk}$ enteric $CH_4$ = daily $CH_4$ production (kg enteric $CH_4$/head.day) as calculated in Eqn 9 above), $N_{ijk}$ = number of dairy cattle in each state (i), stock class (j) and season (k).

Methane from animal waste was estimated using a series of methodologies, algorithms and emission factors from the NGGI report (DCCEE 2009), based on research by Williams (1993) and IPCC (1997) guidelines. Methane production (kg $CH_4$/head.day) from manure management requires the calculation of volatilise solids (VS) excreted/ head.day, based on DM intake and DMD and is calculated as:

$$VS_{ijk} \text{ (kg/head.day)} = I_{ijk} \times (1 - (DMD_{ijk}/100) \times (1 - A) \quad (11)$$

where $I_{ijk}$ = dry matter intake (kg /head.day; as calculated in Eqn 4 above), $DMD_{ijk}$ = dry matter digestibility (g/ 100 g DM) of the diet and A = ash content expressed as a fraction (assumed to be 8% of faecal DM).

From this, daily animal waste $CH_4$ production ($M_{ijk;}$ waste $CH_4$; kg $CH_4$/head.day) is calculated as:

$$M_{ijk} \text{ waste } CH_4 \text{ (kg } CH_4\text{/head.day)} = VS_{ijk} \times B_o \times MCF \times \rho \quad (12)$$

where $VS_{ijk}$ = volatilise solids (kg/head.day; as calculated in Eqn11 above), $B_o$ = emission potential (0.24 $m^3$/ kg VS), MCF = integrated methane conversion factor [%; DCCEE (2009) state-based defaults of 2.75 for WA; 4.57 for QLD & NT; 6.5 for NSW, ACT, TAS & VIC; and 10.07 for SA] and ρ = density of methane (0.662 kg/$m^3$).

From this, total animal waste $CH_4$ production (Gg $CH_4$/annum) is calculated as:

$$\text{Total waste } CH_4 \text{ production (Gg } CH_4\text{/annum)} = \sum (M_{ijk} \text{ waste } CH_4 \times N_{ijk} \times 365) \times 10^{-6} \quad (13)$$

where $M_{ijk}$ waste $CH_4$ = daily $CH_4$ production (kg waste $CH_4$/head.day) as calculated in Eqn 12 above) and $N_{ijk}$ = number of dairy cattle in each state (i), stock class (j) and season (k).

#### *4.2.6. Calculating on-farm nitrous oxide emissions*

Nitrous oxide emissions associated with animal faeces, urine and waste were estimated using methodologies, algorithms and emission factors that reflect Australian conditions (DCCEE 2009), based on research by the Australian Standing Committee on Agriculture (1990) and Freer *et al.* (1997).

The crude protein intake ($CPI_{ijk}$; kg/head.day) is calculated as:

$$CPI_{ijk} \text{ (kg/head.day)} = I_{ijk} \times (CP_{ijk} / 100) \quad (14)$$

where $I_{ijk}$ = DM intake (kg DM/head.day; as calculated in Eqn 4 above) and $CP_{ijk}$ = crude protein (g/ 100 g DM) of the diet.

The amount of N excreted in faeces ($F_{ijk}$; kg/head.day) is calculated as:

$$F_{ijk} \text{ (kg N/ head.day)} = ((0.3 \times (CPI_{ijk} \times (1- ((DMD_{ijk} + 10) / 100)))) + (0.105 \times (ME_{ijk} \times I_{ijk} \times 0.008)) + (0.0152 \times I_{ijk})) / 6.25 \quad (15)$$

where $CPI_{ijk}$ = crude protein intake (kg/head.day) as calculated in Eqn 14 above), $DMD_{ijk}$ = dry matter digestibility (g/ 100 g DM) of the diet, $ME_{ijk}$ = metabolisable energy [MJ/kg DM; calculated as 0.1604 x $DMD_{ijk}$ (g/ 100g DM) – 1.037 (Minson and McDonald 1987)], $I_{ijk}$ = DM intake (kg DM/head.day; as calculated in Eqn 4 above) and 1/6.25 = factor for converting CP into N.

The amount of N that is retained by the animal ($NR_{ijk}$; kg/head.day) in milk and body tissue is calculated as:

$$NR_{ijk} \text{ (kg N/ head.day)} = ((0.032 \times MP_{ijk}) + (0.212 - 0.008 \times (L_{ijk} - 2) - ((0.140 - 0.008 \times (L_{ijk} - 2)) / (1 + \exp(-6 \times (Z_{ijk} - 0.4))))) \times (LWG_{ijk} \times 0.92)) / 6.2 \quad (16)$$

where $MP_{ijk}$ = milk production (kg/head.day), $L_{ijk}$ = intake relative to maintenance (as calculated in Eqn 7 above), $Z_{ijk}$ = relative size (live weight / standard reference weight for each stock class; available in DCCEE 2009), $LWG_{ijk}$ = live weight gain (kg/day) and 1/6.25 = factor for converting CP into N.

The amount of N excreted in urine ($U_{ijk}$; kg/head.day) is calculated as:

$$U_{ijk} \text{ (kg N/ head.day)} = (CPI_{ijk} / 6.25) - F_{ijk} - NR_{ijk} - ((1.1 \times 10^{-4} \times W_{ijk}^{0.75}) / 6.25) \quad (17)$$

where $CPI_{ijk}$ = crude protein intake (kg/head.day; as calculated in Eqn14 above), $F_{ijk}$ = N excreted in faeces (kg N/ head.day; as calculated in Eqn 15 above), $NR_{ijk}$ = N retained by the animal (kg N/ head.day; as calculated in Eqn 16 above), $W_{ijk}$ = live weight (kg/head) and 1/6.25 = factor for converting CP into N.

Total faeces ($AF_{ijk}$; Gg N/annum) and total urinary ($AU_{ijk}$; Gg N/annum) N excreted is calculated as:

$$AF_{ijk} \text{ (Gg N/annum)} = \sum F_{ijk} \times N_{ijk} \times 365 \times 10^{-6} \quad (18)$$

$$AU_{ijk}\ (\text{Gg N/annum}) = \sum U_{ijk} \times N_{ijk} \times 365 \times 10^{-6} \quad (19)$$

where $F_{ijk}$ = N excreted in faeces (kg N/ head.day; as calculated in Eqn 15 above), $U_{ijk}$ = N excreted in urine (kg N/ head.day; as calculated in Eqn 17 above) and $N_{ijk}$ = number of dairy cattle in each state (i), stock class (j) and season (k).

The direct and indirect $N_2O$ emissions from faeces and urine voided onto pastures directly and from stored/spread faeces and urine is estimated using the total faeces ($AF_{ijk}$) and total urine ($AU_{ijk}$) figures derived from Eqns 18 and 19, respectively, with the emission factors and equations presented in Table 4.4. The DCCEE (2009) methodology defines the percentage of faeces and urine allocated to one of four manure management systems depending on the location of the farm. In New South Wales, Australian Capital Territory, Tasmania and Victoria, 92% of annual faeces and urine is deposited onto pastures during grazing, 6% is stored in a lagoon system, 1.5% is spread daily, and 0.5% is stored as a liquid/slurry. In Queensland, 90% of annual faeces and urine is deposited onto pastures during grazing, 7% is spread daily and 3% is stored in a lagoon system. In Western Australia, 92% of annual faeces and urine is deposited onto pastures during grazing, 6% is spread daily and 2% is stored in a lagoon system. In South Australia, 88.5% of annual faeces and urine is deposited onto pastures during grazing, 10% is stored in a lagoon system, 1% is spread daily and 0.5% is stored as a liquid/slurry.

Nitrous oxide emissions associated with N fertilisers were estimated using methodologies, algorithms and emissions factors (Table 4.4) that reflect Australian conditions based on research by Galbally *et al.* (2005). The study did not differentiate between N fertiliser applied to pastures or crops, and given the slightly higher emission factor for pastures compared with crops (0.004 cf. 0.003, respectively), this study assumed that all N fertiliser was applied to pasture.

**Table 4.4** Emission factors and equations to estimate direct and indirect nitrous oxide emissions from faeces, urine, stored and spread waste and nitrogen fertilisers (DCCEE 2009).

| | Source | Equation and emission factors to estimate $N_2O$ losses |
|---|---|---|
| Direct | Faeces excreted onto pastures | 0.005 x faeces N x % faeces deposited onto pastures during grazing x 1.57 |
| | Urine excreted onto pastures | 0.004 x urinary N x % urine deposited onto pastures during grazing x 1.57 |
| | Stored waste | 0.001 x sum of faeces & urinary N x % faeces and urinary N stored in lagoons and as liquid/slurry[A] x 1.57 |
| | Spread stored waste | 0.01 x (faeces & urinary N x % of faeces & urinary N stored – $N_2O$ lost during the storage phase – $N_2O$ lost through volatilisation) x 1.57 |
| | N fertiliser applications | ((0.004 x N fertiliser applied to pastures) + (0.003 x N fertiliser applied to crops)) x 1.57 |
| Indirect | Volatilisation (faeces and urine) | (0.01 x ((% faeces & urinary N deposited onto pastures x 0.2) + (% faeces & urinary N stored in lagoon x 0.35) + (% faeces & urinary N stored as liquid/slurry x 0.4) + (% faeces & urinary N spread daily x 0.07))) x 1.57 |
| | Volatilisation (N fertiliser) | 0.1 x 0.01 x sum N in fertiliser applied to pastures & crops x 1.57 |
| | Leaching/runoff (faeces and urine) | 0.3 x 0.0125 x (faeces N + urinary N + spread and stored waste N) x 1.57 |
| | Leaching/runoff (fertiliser) | 0.3 x 0.0125 x sum N in fertiliser applied to pastures & crops x 1.57 |

[A] Faeces and urine stored and spread daily is also classified as stored waste: however, this source of waste does not emit $N_2O$ during the storage phase, only the spreading phase, and therefore is not a source of stored $N_2O$ emissions.

### *4.2.7. Farming classification*

Farms were classified according to their FS as described by Dairy Australia (2011a). The FS classification is defined as FS1 [grazed pasture year-round with supplementary forage fed in paddocks and low grain feeding (< 1 t DM/cow.lactation)], FS2 [grazed pasture year-round, with supplementary forage fed in paddocks and medium to high grain feeding (> 1 t DM/cow.lactation)], FS3 (grazed pasture year-round with supplementary forages and other feeds fed as a partial mixed ration on a feedpad as required and low to high grain feeding), FS4 (grazed pastures for < 9 months of the year with a partial mixed ration fed on a feedpad area as required and low to high grain feeding) and FS5 (zero grazing of milking herd, fed total mixed ration year round and generally housed indoors). This study consisted of 11 FS1 farms, 20 FS2 farms and 10 FS3 farms. While this study did not assess the GHG emissions of farms classified as either FS4 or FS5, nationally less than 10% of the farms are identified as being FS4 or FS5 (Dairy Australia 2011b), therefore supporting the conclusion that the results from this study are reflective of the majority of Australian dairy farms.

#### *4.2.8. Statistical analyses*

Statistical Program for the Social Sciences Statistics (SPSS 2008) was used to for all statistical data analysis. Multiple regression analysis was used to describe the influence of annual milk production, milking herd size and total farm area on total farm GHG emissions. An SMLR analysis between GHG emissions intensities and individual key farm variables was undertaken using the farm variables of milk production per cow (kg FPCM/cow), milk production per ha (t FPCM/ha), stocking rate (number of milkers/ha of milking platform), pasture consumption (t DM consumed/ha), total feed intake (t DM/cow.lactation), feed conversion efficiency (kg of FPCM/kg DM intake), proportion of grain in the milking herd diet and N fertiliser application rate (kg N/ha). The influence of FS and region on the GHG emissions intensity of milk production (kg $CO_2$e/kg FPCM), cow intensity (t $CO_2$e/cow) and farm area intensity (t $CO_2$e/ha) were analysed separately using a one-way ANOVA procedure. In addition, following testing the data for normality, a cumulative distribution function of the GHG emissions intensity of milk production for each FS was constructed using the NORMDIST (value, mean, standard deviation, TRUE) function in Microsoft Excel 2007 (Microsoft Corporation 2007).

### 4.3 Results

#### *4.3.1. Farm greenhouse gas emissions*

The mean estimated GHG emissions intensity of milk production was 1.04 ± 0.17 kg $CO_2$e/kg FPCM. The mean estimated GHG emissions intensity of milk production for Tasmania was 1.30 kg $CO_2$e/kg FPCM, which was significantly ($P < 0.05$) higher than all other regions, with the exception of Queensland (Table 4.5). The mean estimated GHG emissions intensity per cow was 6.34 ± 0.77 t $CO_2$e/cow.annum, with no significant ($P > 0.05$) regional differences (Table 4.5). The mean estimated GHG emissions intensity per hectare was 7.74 ± 3.80 t $CO_2$e/ha.annum, with Tasmania and south eastern Victoria being significantly ($P < 0.05$) higher than New South Wales, Western Australia and Queensland (Table 4.5).

There was a positive linear relationship between total farm GHG emissions and either annual milk production or milking herd size for the whole dataset as shown by the high coefficient of determination in Eqns 20 and 21. Therefore, at whole of industry assessment, milk production or number of milking cows could be used as a suitable surrogate for estimating total GHG emissions. However, on an individual farm basis, the GHG emissions intensity of milk production varied between 0.76 and 1.68 kg $CO_2$e/kg FPCM (Figure 4.1a) while the GHG emissions intensity per cow also varied between 4.78 and 8.59 t $CO_2$e/cow (Figure 4.1b). This substantial variation between farms reduces the acceptability of a single emission factor (milk production or milking cow number) to be used as a surrogate for quantifying on farm emissions. Area was not a suitable surrogate for estimating total GHG emissions as shown by low coefficient of determination in Eqn 22 and the large variation in GHG emissions intensity per hectare (Figure 4.1c).

Total farm GHG emissions (t $CO_2$e/annum) = 0.89 x annual milk production (t FPCM) + 258.34; $R^2$ = 0.95 (20)

Total farm GHG emissions (t $CO_2$e/annum) = 6.46 x milking herd size – 41.81; $R^2$ = 0.97 (21)

Total farm GHG emissions (t $CO_2$e/annum) = 3.97 x total farm area (ha) + 911.73; $R^2$ = 0.30 (22)

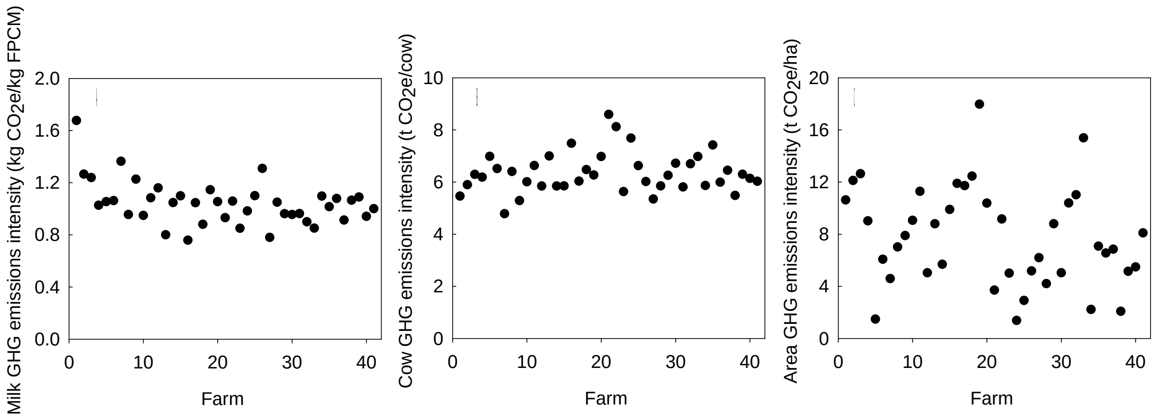


**Figure 4.1** The greenhouse gas emissions intensity of milk production (kg $CO_2e$/kg fat and protein corrected milk; A), per cow (t $CO_2e$/cow; B) and per hectare (t $CO_2e$/ha; C) for individual farms.

The contribution of the various GHG emission sources, as a percentage of total farm GHG emissions, for each state is presented in Table 4.6. Enteric $CH_4$ was the biggest source of total farm GHG emissions, with an overall mean of 55.5%, with regional means varying between 49.8 and 57.8% (Table 4.6). On-farm $CO_2$ from electricity and diesel consumption and indirect $N_2O$ emissions from animal waste, at 9.6 and 8.4%, respectively, were the next two largest sources (Table 4.6).

**Table 4.5** Regional means and ranges of total farm greenhouse gas emissions (t $CO_2$e/annum) and greenhouse gas emissions intensities [kg $CO_2$e/kg fat and protein corrected milk (FPCM); t $CO_2$e/cow.annum; t $CO_2$e/ha].

| | Number of farms | Total farm GHG emissions t $CO_2$e/annum | | Greenhouse gas emissions intensity kg $CO_2$e/kg FPCM | | t $CO_2$e/cow.annum | | t $CO_2$e/ha.annum | |
|---|---|---|---|---|---|---|---|---|---|
| | | **Mean** | **Range** | **Mean** | **Range** | **Mean** | **Range** | **Mean** | **Range** |
| NSW | 9 | 1723[bc] | 411 – 3,95 | 1.06[b] | 0.85 – 1.31 | 6.77[a] | 5.29 – 8.59 | 5.96[b] | 1.38 – 10.38 |
| QLD | 4 | 1184[c] | 621 – 1,63 | 1.11[ab] | 0.95 – 1.36 | 6.17[a] | 4.78 – 6.98 | 4.79[b] | 1.48 – 7.01 |
| SA | 3 | 4450[a] | 1167 – 9416 | 0.99[b] | 0.85 – 1.10 | 6.75[a] | 5.86 – 7.42 | 8.23[ab] | 2.22 – 15.39 |
| TAS | 4 | 3645[ab] | 2240 – 5,28 | 1.30[a] | 1.03 – 1.68 | 5.96[a] | 5.45 – 6.29 | 11.09[a] | 9.01 – 12.63 |
| Nth VIC | 3 | 2521[abc] | 4121 – 4770 | 0.94[b] | 0.90 – 0.96 | 6.41[a] | 5.81 – 6.72 | 8.81[ab] | 5.03 – 11.01 |
| SE VIC | 10 | 1993[bc] | 742 – 5505 | 1.00[b] | 0.76 – 1.16 | 6.34[a] | 5.84 – 7.48 | 10.38[a] | 5.03 – 17.97 |
| SW VIC | 3 | 1639[bc] | 883 – 3014 | 0.93[b] | 0.78 – 1.05 | 5.82[a] | 5.34 – 6.25 | 6.39[ab] | 4.20 – 8.79 |
| WA | 5 | 2373[abc] | 713 – 4602 | 1.02[b] | 0.91 – 1.09 | 6.07[a] | 5.48 – 6.45 | 5.22[b] | 2.07 – 6.84 |
| Mean | 41 | 2255 | 411 – 9416 | 1.04 | 0.76 – 1.68 | 6.34 | 4.78 – 8.59 | 7.74 | 1.38 – 17.97 |

Superscript letters that differ within columns indicate values that are significantly different at $P = 0.05$

**Table 4.6** Percentage (%) of total farm greenhouse gas emissions from each source for each dairy region, as estimated using the DGAS calculator.

| **Source of greenhouse gas emission** | **NSW** | **QLD** | **SA** | **TAS** | **Nth VIC** | **SE VIC** | **SW VIC** | **WA** | **Mean** |
|---|---|---|---|---|---|---|---|---|---|
| Enteric $CH_4$ (%) | 54.4 | 56.6 | 49.8 | 55.3 | 55.2 | 56.4 | 56.2 | 57.8 | 55.5 |
| $CO_2$ from fuel & electricity (%) | 11.5 | 10.8 | 12.7 | 10.9 | 8.6 | 8.8 | 8.1 | 5.5 | 9.6 |
| Indirect $N_2O$ from animal waste[a] (%) | 8.2 | 7.5 | 7.5 | 9.3 | 7.6 | 8.9 | 7.6 | 9.2 | 8.4 |
| Direct $N_2O$ from animal waste[a] (%) | 6.4 | 6.0 | 5.9 | 7.2 | 5.9 | 6.9 | 5.9 | 7.4 | 6.6 |
| $CO_2$ from purchased grains & concentrates (%) | 6.8 | 7.2 | 6.9 | 2.3 | 5.6 | 5.3 | 8.7 | 6.1 | 6.0 |
| $CH_4$ from animal waste (%) | 5.1 | 3.8 | 7.1 | 4.7 | 4.8 | 5.1 | 5.3 | 2.3 | 4.7 |
| $CO_2$ from purchased fertilisers (%) | 1.9 | 3.0 | 2.4 | 4.0 | 1.0 | 3.0 | 3.2 | 5.0 | 2.9 |
| $CO_2$ from purchased forage (%) | 3.0 | 0.1 | 5.0 | 1.2 | 10.1 | 1.6 | 0.9 | 0.5 | 2.4 |
| Direct $N_2O$ from N fertilisers (%) | 1.4 | 2.8 | 1.5 | 2.7 | 0.7 | 2.2 | 2.3 | 3.3 | 2.1 |
| Direct $N_2O$ from N fertilisers (%) | 1.2 | 2.3 | 1.3 | 2.3 | 0.6 | 1.9 | 1.9 | 2.8 | 1.8 |

[a] Includes faeces and urine voided directly onto pastures during grazing and faeces and urine stored and spread onto pastures either daily or at a later time

### *4.3.2. Stepwise multiple linear regression analyses*

The SMLR analysis showed that milk production per cow (kg FPCM/cow.lactation) was the only significant key farm variable influencing the GHG emissions intensity of milk production (kg $CO_2$e/kg FPCM) and accounted for 0.70 of the variation (Table 4.7). The SMLR analysis showed that milk production per cow (kg $CO_2$e/kg FPCM) alone could explain 0.64 of the variation in emissions intensity per cow (t $CO_2$e/cow.annum). The addition of percentage of the milking herds' diet consisting of grain to the model could only account of an additional 0.04 of the variation (Table 4.7). The SMLR analysis showed that milk production per hectare (t FPCM/ha.annum) alone could explain 0.88 of the variation in GHG emissions intensity per unit area (t $CO_2$e/ha.annum). The addition of milk production per cow (kg FPCM/cow.lactation) and N fertiliser application rate (kg N/ha.annum) could only account for an additional 0.09 of the variation (Table 4.7). Milk production per cow was the only common variable influencing the three intensities, with increased milk production per cow decreasing milk and area GHG emissions intensity, while it increased the cow GHG emissions intensity (Table 4.7).

**Table 4.7** Models of stepwise multiple linear regression of the greenhouse gas emissions intensity expressed as milk intensity (kg $CO_2e$/kg fat and protein corrected milk; FPCM), cow intensity (t $CO_2e$/cow.annum) and area intensity (t $CO_2e$/ha.annum), where *b* is the unstandardized coefficient, *SE b* is the standard error of *b*, *β* is the standardized coefficient and $R^2$ is the coefficient of determination.

| **Milk intensity (kg $CO_2e$/kg FPCM)** | | ***b*** | ***SE b*** | ***β*** | **$R^2$** |
|---|---|---|---|---|---|
| Step 1 | Constant | 1.685 | 0.069 | | 0.698 |
| | Milk production (kg FPCM/cow.lactation) | -1.0E-04 | -1.1E-05 | -0.835*** | |
| **Cow intensity (t $CO_2e$/cow.annum)** | | ***b*** | ***SE b*** | ***β*** | **$R^2$** |
| Step 1 | Constant | 3.572 | 0.342 | | 0.639 |
| | Milk production (kg FPCM/cow.lactation) | 4.4E-04 | 5.3E-05 | 0.799*** | |
| Step 2 | Constant | 3.633 | 0.326 | | 0.682 |
| | Milk production (kg FPCM/cow.lactation) | 3.8E-04 | 5.9E-05 | 0.677*** | |
| | Grain feeding (%)[a] | 0.016 | 7.2E-03 | 0.240* | |
| **Area intensity (t $CO_2e$/ha.annum)** | | ***b*** | ***SE b*** | ***β*** | **$R^2$** |
| Step 1 | Constant | 0.974 | 0.461 | | 0.875 |
| | Milk production (t FPCM/ha.year) | 0.887 | 0.054 | 0.935*** | |
| Step 2 | Constant | 5.300 | 0.631 | | 0.951 |
| | Milk production ((t FPCM/ha.year) | 0.985 | 0.036 | 1.038*** | |
| | Milk production (kg FPCM/cow.lactation) | -8.1E-04 | 1.1E-04 | -0.295*** | |
| Step 3 | Constant | 4.581 | 0.593 | | 0.963 |
| | Milk production (t FPCM/ha.year) | 0.919 | 0.037 | 0.969*** | |
| | Milk production (kg FPCM/cow.lactation) | -7.0E0.4 | 9.7E-05 | -0.256*** | |
| | Nitrogen fertiliser (kg N/ha.year) | 7.3E-03 | 2.1E-03 | 0.128** | |

[a] % grain in the milker diet

Significant contributions to the model at $P < 0.05$ (*), $P < 0.01$ (**); $P < 0.001$ (***)

#### *4.3.3. Influence of farming system on greenhouse gas emissions intensity*

The FS1 group exhibited a significantly ($P < 0.05$) higher GHG emissions intensity of milk production, at 1.23 kg $CO_2$e/kg FPCM, compared with the FS2 and FS3 groups, at 0.98 and 0.97 kg $CO_2$e/kg FPCM, respectively (Table 4.8). The FS2 group exhibited a significantly ($P < 0.05$) higher GHG emissions intensity per cow, at 6.78 t $CO_2$e/cow.annum, compared with the FS1 and FS3 groups, at 5.79 and 6.08 t $CO_2$e/cow.annum, respectively (Table 4.8). There was no significant ($P > 0.05$) difference in GHG emissions intensity per unit area, at 8.21, 7.67 and 7.37 t $CO_2$e/ha.annum for the FS1, FS2 and FS3 groups, respectively (Table 4.8).

**Table 4.8** The mean greenhouse gas emissions intensity [kg $CO_2$e/kg fat and protein corrected milk (FPCM); t $CO_2$e/cow.annum; t $CO_2$e/ha.annum] for each farming system group.

| | Greenhouse gas emissions intensity | | |
|---|---|---|---|
| | **kg $CO_2$e/kg FPCM** | **t $CO_2$e/cow.annum** | **t $CO_2$e/ha.annum** |
| FS1 | 1.23[a] | 5.79[b] | 8.21[a] |
| FS2 | 0.98[b] | 6.78[a] | 7.67[a] |
| FS3 | 0.97[b] | 6.08[b] | 7.37[a] |

(FS1 = pasture based with low grain feeding; FS2 = pasture-based with medium to high grain feeding and supplements fed in grazed paddocks; FS3 = pasture-based with low to high grain feeding and supplements fed on feedpad area as required).

Superscript letters that differ within columns indicate values that are significantly different at P = 0.05

The cumulative distribution function of the GHG emissions intensity of milk production for the three FS groups showed little variation between FS2 and FS3, with 95% of the farms in FS2 having a GHG emission intensity of milk production between 0.77 and 1.12 kg $CO_2$e/kg FPCM compared with between 0.86 and 1.09 kg $CO_2$e/kg FPCM for the FS3 group (Figure 4.2). In contrast, there was a substantially higher variation for FS1, with 95% of the FS1 farms having a GHG emissions intensity of milk production between 1.04 and 1.60 kg $CO_2$e/kg FPCM (Figure 4.2).

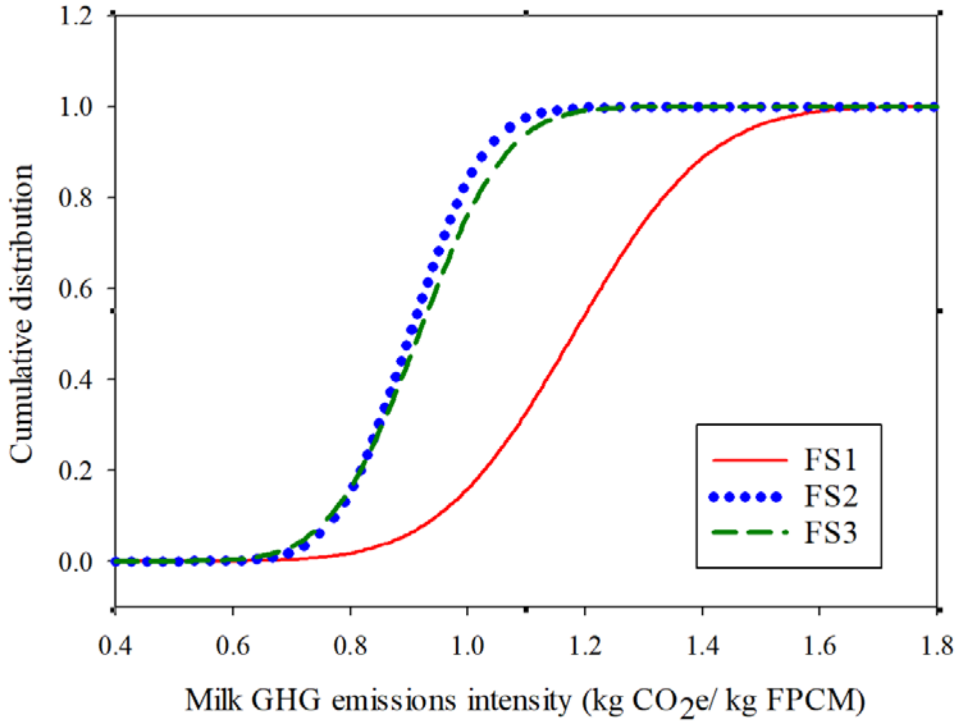


**Figure 4.2** Cumulative distribution function of the greenhouse gas emissions intensity of milk production (kg $CO_2$e/kg fat and protein corrected milk) for the three farming systems groups (FS1 = pasture based with low grain feeding (solid red line); FS2 = pasture-based with medium to high grain feeding and supplements fed in grazed paddocks (dotted blue line); FS3 = pasture-based with low to high grain feeding and supplements fed on feedpad areas as required (dashed green line)).

## 4.4 Discussion

To date, few studies have been undertaken to estimate the GHG emissions associated with dairy production in Australia. This study was unique in that farms were selected from throughout all dairying regions of Australia, as opposed to a single region (English 2007; Christie *et al.* 2011). In addition, actual farm data, as opposed to hypothetical data was used (Basset-Mens *et al.* 2005; Browne *et al.* 2011), and farms were selected across a range of FS varying from predominantly pasture- based with no or low grain supplement through to relatively high levels of grain inputs. In addition accurate seasonal feed quality values, as opposed to annual average ‘textbook’ values were used to determine the diet quality parameters (Beukes *et al.* 2011).

In assessing the GHG emissions of 41 Australian dairy farms, total annual milk production was shown to account for 95% of the variation in the estimated total farm GHG emissions. Given that the inventory assessment used in this study incorporates milk production into the equations for estimating enteric $CH_4$ emissions and that enteric $CH_4$ was the largest source of emissions, it is not surprising that such a correlation was found. In experimental studies measuring daily intakes, enteric $CH_4$ production and milk production, the positive relationship between enteric $CH_4$ emission production and milk production per cow has been shown (Ulyatt *et al.* 2002a, 2002b; Lovett *et al.* 2005; O'Neill *et al.* 2011). Boadi *et al.* (2004), in reviewing several studies, reported the emission intensity of milk production varied between 11.4 and 28.3 L $CH_4$/kg milk; equivalent to between 0.31 and 0.76 kg $CO_2$e/kg milk from enteric $CH_4$. The results of this study were not dissimilar to that of the Boadi *et al.* (2004) review, varying between 0.39 and 0.88 kg $CO_2$e/kg FPCM from enteric $CH_4$. It is also important to note that this included enteric $CH_4$ emissions from all stock, not just the milking herd.

It is clear that while the relationship between total milk production and total farm GHG emissions suggests that milk production alone could be a suitable surrogate for estimating GHG emissions for national inventory purposes, using a single emissions factor, such as total annual milk production, to estimate any given individual farm's GHG emissions, has the potential to either substantially under or overestimate individual farms' GHG emissions. When total annual milk production was used with Eqn 20 to estimate total farm GHG emissions, only 18 of the 41 farms' total farm GHG emissions was within 10% of their DGAS-estimate, highlighting that the use of a single emission factor like milk production is not a suitable surrogate for individual farms GHG estimations. At the two extremes, one farm's total farm GHG emissions was underestimated by 30% while another farm's total farm GHG emissions was overestimated by 41%. In addition, the GHG emissions intensity of milk production, on an individual farm basis, varied between 0.76 and 1.68 kg $CO_2$e/kg FPCM. Exploring reasons as to the variation in the GHG emissions intensity of milk production is critical and may assist in exploring potential mitigation strategies for maintaining total farm GHG emissions while increasing total annual milk production.

Results from this study were congruent to studies from other countries. In a study comparing 10 dairy farms in Ireland, Casey and Holden (2005a) reported a range of between 0.92 and 1.51 kg $CO_2$e/kg milk, while Basset-Mens *et al.* (2005) found that the typical New Zealand dairy farm emitted 0.72 kg $CO_2$e/kg milk. Results from 23 conventional and organic farms in Sweden ranged between 0.90 and 1.04 kg $CO_2$e/kg milk (Cederberg and Flysjö 2004), while results from Germany comparing 18 farms found that the GHG emissions ranged between 1.0 and 1.3 kg $CO_2$e/kg milk (Haas *et al.* 2001). In two more recent studies, the GHG emissions of Oceania (dominated by the Australian and New Zealand dairy industries) was estimated at ~ 1.1 – 1.2 kg $CO_2$e/kg milk (FAO 2010; Hagemann *et al.* 2011), reaffirming that results from our study were comparative to other international studies. However, comparing the results of this study with results from other studies can be difficult given the impact that different methodologies, emissions factors and assumptions can have on the estimations. For example, the direct $N_2O$ emissions from N fertilisers applied to pastures is 0.4% in Australia (DCCEE 2009). This is considerably lower than the IPCC emission factor of 1.25% as is used in many European studies (e.g. Casey and Holden 2005a; Lovett *et al.* 2006) or 1.0% as is used in New Zealand studies (e.g. Beukes *et al.* 2011; Flysjö *et al.* 2011b). The direct $N_2O$ emission factors for animal waste are also lower in Australia, at 0.4 and 0.5% for urinary and faecal deposition onto pastures, respectively (DCCEE 2009), compared with the default IPCC emission factor of 2.0% (IPCC 2006), with the result that direct $N_2O$ emissions will be lower than indirect $N_2O$ emissions for Australian dairy GHG emission studies when compared with other international studies.

There was a significant ($P < 0.05$) regional difference in the GHG emissions intensity of milk production, however, caution needs to be taken when extrapolating the result of the small number of farms in this study to the whole of industry for any particular region. The mean GHG emissions intensity of milk production for the four Tasmanian dairy farms in this study was 1.30 kg $CO_2$e/kg FPCM compared with a mean of 1.04 kg $CO_2$e/kg FPCM when 60 Tasmanian dairy farms were assessed for their GHG emissions intensity (Christie *et al.* 2011). All four Tasmanian farms in this study had low levels of grain feeding (mean of 0.46 t DM/cow.lactation compared with an overall study mean of 1.29 t DM/cow.lactation), exhibited low milk production per cow (mean of 4329 kg FPCM/cow.lactation compared with an overall study mean of 6265 kg FPCM/cow.lactation) and were classified as FS1. Based on a recent survey of 80 Tasmanian dairy farm operators (Dairy Australia

2011b), 66% of farms were identified as FS1, 29% as FS2 and 5% as FS3. If one or two of the four Tasmanian farms in this study were an FS2 or FS3 as opposed to FS1, it is possible that the Tasmanian mean GHG emissions intensity of milk production may have been lower. For this reason, a comparisons of the FS across all regions, was considered the preferable method than a regional comparison.

The cumulative distribution function for the three FS showed substantially wider variation in the GHG emissions intensity of milk production for the FS1 farms compared with substantially less variation between the FS2 and FS3 farms. One of the major differences between the three FS was in the level of milk production per cow, with the FS1 group producing on average 4823 ± 902 kg FPCM/cow.lactation compared with 7055 ± 1,241 and 6271 ± 654 kg FPCM/cow.lactation for the FS2 and FS3 groups, respectively. Given that the allocation of farms to FS classifications was partially based on the level of grain feeding, grain feeding was always lower for the FS1 group with a mean of 0.62 t DM/cow.lactation compared with a mean of 1.78 and 1.06 t DM/cow.lactation for the FS2 and FS3 groups, respectively.

It is well established that increasing the level of grain/concentrate in the diet improves milk production (Tessmann *et al.* 1991; Kellaway and Porta 1993; Robaina *et al.* 1998; Stockdale 1999). In addition, it is also well established that increasing the proportion of grain/concentrate in the diet reduces the proportion of dietary energy converted into $CH_4$ (Moe and Tyrrell 1979; Johnson and Johnson 1995; Boadi *et al.* 2004) and reduces enteric $CH_4$ emissions per unit of milk production (Johnson *et al.* 2002; Lovett *et al.* 2005, 2006). In addition, improving milk production per cow was found to be the only significant key farm variable in the SMLR analysis to influence the GHG emissions intensity of milk production, with a reduction of 0.102 kg $CO_2$e for every additional 1000 kg of FPCM produced per cow. Therefore, it is clear from this study that management practices that increase milk production per cow will reduce the GHG emissions intensity of milk production and that this is a key target area for lowering the emissions intensity of milk production for the Australian dairy industry. However, focusing on improving milk production per cow is likely to result in higher milk production per farm, unless stocking rates are adjusted accordingly to produce similar levels of milk production from fewer animals, It is also important to note that increasing the consumption of home grown forage per hectare, and not milk production per cow, has been shown to be a strong determinant of business success in grazing-based dairy production systems (O'Brien 1994; Savage and Lewis 2005; Chapman *et al.* 2008a, 2009).

While there was no significant ($P > 0.05$) difference in the GHG emissions intensity per unit of area across the three FS, there were significant ($P < 0.05$) regional differences. Tasmania and south-eastern Victoria were significantly ($P < 0.05$) higher in farm area GHG emissions than New South Wales, Western Australia and Queensland. When farms were ranked according to stocking rate (i.e. number of milkers per hectare of milking area), 10 of the highest 15 farms were located either in Tasmania or south-eastern Victoria. As stocking rate increases, there is greater $CH_4$ production per unit of land, thus resulting in higher farm area GHG emissions figures. Some of the lowest stocking rates were in New South Wales, Western Australia and Queensland, further confirming that even though stocking rate was not identified as one of the key farm variables in the area GHG emissions intensity SMLR analysis, stocking rate still appears to be a contributing factor when comparing regional average farm area GHG emissions.

The empirical methodologies used in this study are the only currently IPCC acceptable methods to account for farm GHG emissions at a regional and national scale. However, these emissions can only be considered as an estimate. Given that over half of all emissions were derived from enteric $CH_4$, any variation in the methodology used to calculate this source of emission is likely to have the biggest influence on total farm emissions. The Australian methodology for estimating $CH_4$ emissions use a Blaxter and Clapperton (1965) derived equation, using herd liveweight, daily liveweight gain (for growing stock), diet DMD and milk production figures. In this study using farm and seasonal-specific, laboratory derived feed quality data was a vast improvement for estimating GHG emissions, compared with using potentially inaccurate generic 'textbook' averages. However, these were snap-shot assessments of the diet quality on the day that each farm was visited and as such may not accurately reflect the diet quality for the milking herd for each season or more importantly, for the whole study period.

It is also important to note that there are potentially seasonal influences on $CH_4$ emissions from pastures with similar feed quality. In a study by Ulyatt *et al.* (2002a), sheep were fed a diet with a DMD of 82.0% in mid spring (September) and mid winter (June). Methane emissions varied between the two seasons at 30.6 and 27.9 g

$CH_4$/day, respectively. When converted to digestible DM intake (DDMI) to remove the variation in daily feed intakes between the two study periods, the results were 24.7 and 18.5 g $CH_4$/kg DDMI, respectively. In the same Ulyatt *et al.* (2002a) study, dairy cows were fed a diet with 82% DMD in early spring, resulting in a $CH_4$ emission of 27.3 g $CH_4$/kg DDMI. Even when the diet quality for the dairy cow study was reduced to 75.5 and 68.4% DMD in late spring (November) and early autumn (March), respectively, $CH_4$ emissions were not significantly ($P<0.05$) different at 18.2 and 18.0 g $CH_4$/kg DDMI. The Ulyatt *et al.* (2002a) study showed that diets with the same DMD% resulted in varied $CH_4$ emissions both within and between ruminant species. However it is important to know what other contributing factors, other than DMD, could have resulted in differing enteric $CH_4$ production. If these factors can be identified and incorporated into our current methodologies, this would assist in strengthening the accuracy of enteric $CH_4$ emission estimations.

Palliser and Woodward (2002) compared measured $CH_4$ emissions from lactating dairy cows (Woodward *et al.* 2002) with estimated $CH_4$ emissions from one mechanistic (Baldwin 1995) and three empirical models (Blaxter and Clapperton 1965; Moe and Tyrrell 1979; Kirchgessner *et al.* 1995). They found that the empirical Blaxter and Clapperton (1965) model consistently over predicted $CH_4$ emissions. While the mechanistic model was found to be a better estimate of $CH_4$ emissions, variations between measured and predicted $CH_4$ emissions were still present with this model (Palliser and Woodward 2002).

Ellis *et al.* (2010) further confirmed this when they compared the observed $CH_4$ emissions from 206 data points derived from 16 different studies with the estimated $CH_4$ emissions from nine $CH_4$ prediction equations. These nine $CH_4$ emissions equations varied in their level of detail required, with some only needing daily gross energy intake to estimate $CH_4$ emissions [e.g. the IPCC (1997) Tier II equation]while others required substantially greater data to estimate $CH_4$ [e.g. the Moe and Tyrrell (1979) equation requires non-structural carbohydrate, hemicellulose and cellulose figures). The general conclusions drawn from the authors was that while some equations predict $CH_4$ emissions better than others, all equations had some degree of difficulty describing the variation present in observed $CH_4$ values and prediction accuracy appeared to be low.

Although agriculture is currently excluded from the carbon tax (or the subsequent emissions trading scheme) that the current Australian government is legislating (DCCEE 2011b), agriculture is considered an important component in meeting Australia's GHG emission targets. To facilitate this, the Australian government has legislated the Carbon Farming Initiative (DCCEE 2011c) to provide a mechanism and financial incentive to assist agriculture in adopting practices that can provide emission offsets in one of two ways; by removing or avoiding emissions (e.g. the capture and destruction, or abatement of enteric $CH_4$ from livestock) or by removing carbon from the atmosphere and storing it in trees or soil (e.g. farming in a manner to increase soil carbon). Collecting accurate on-farm information so as to undertake a 'business-as-usual' GHG emissions assessment will be the critical first step in this process. This study has shown significant variation in GHG emissions intensity of milk production exists between and within FS and as such, a single emission factor for milk production is not appropriate for estimating total farm emissions. It is also apparent that the current Australian inventory methodology for estimating GHG emissions may have some limitations, although finding the balance between simplicity of data collection and overall accuracy of emissions estimation will continue to be an issue given that on-farm emission measuring is unlikely to ever be practical. However, with on-going field research validating and improving the algorithms and emission factors currently used to estimate Australian dairy GHG emissions, this will further strengthen our ability to estimate on-farm GHG emissions.

While identifying and implementing mitigation strategies to reduce the GHG emissions intensity of milk production is important, it is critical that any mitigation strategy is also well adapted to alleviate potential climate change influences. Similarly, the dairy industry also needs be mindful of any maladaptation to climate change, where the most likely adaptation strategies to climate change could also result in increasing levels of GHG emissions. Climate change projections for Australia indicate increasing temperatures, changing rainfall patterns, and elevated atmospheric $CO_2$ concentrations will occur into the future (CSIRO and BoM 2007). Modelling of the resistance of pasture production to incremental changes in daily temperatures, rainfall patterns and atmospheric $CO_2$ levels has shown that these changes may have a positive impact on annual pasture production for some

Australian dairy regions, predominantly as a result of increased winter temperatures and atmospheric $CO_2$ concentrations (e.g. north west Tasmania; Cullen *et al.* 2009, 2012). Adaptation strategies to capture this increased pasture production could include an increase in stocking rate and/ or lowering the rate of concentrate feeding (Cullen *et al.* 2010). According to this study, the result of implementing these adaptation strategies would be an increase in the GHG emissions intensity of milk production. Therefore, there is also an emerging conflict between how dairy farms may adapt to a changing climate and mitigating the GHG emissions associated with milk production in Australia.

### 4.5 Conclusion

The work presented in this paper is the first known study that explores the GHG emissions of differing farm systems encompassing all dairy regions of Australia, varying intensities of milk production from both a per-cow and per-hectare basis and varying reliance on inputs such supplementary feeding, fertiliser and irrigation. The results of this study indicated that adopting a more intensive dairy FS, with higher inputs from grain and other supplements to increase milk production per cow, resulted in reducing the GHG emissions intensity of milk production. However, the Australian dairy industry has traditionally focussed its dairy systems research efforts on the production and consumption of home grown forage, as this has been shown to be a key driver of business success. If farmers are going to adopt strategies that focus on improving per cow production through a greater reliance on off-farm supplements, they need to be aware of the financial and management implications of this at whole of farm level, above and beyond the GHG emission implications.

### 4.6 Acknowledgements

This study was supported by funding from Dairy Australia, the Tasmanian Institute of Agriculture and the Australian Government Department of Agriculture, Fisheries and Forestry under its Australia’s Farming Future Climate Change Research Program. The authors want to thank Dairy Australia, The Department of Primary Industries Victoria, Incitec-Pivot, Megafert, Impact and Land and Water Australia for funding the A4N program. In addition the authors would like to thank the farmers who participated in the A4N program and acknowledge the team of data collectors and those associated with collating, processing and interpreting farm data.

# CHAPTER 5 USING A MODELLING APPROACH TO EVALUATE TWO OPTIONS FOR IMPROVING ANIMAL NITROGEN USE EFFICIENCY AND REDUCING NITROUS OXIDE EMISSIONS ON DAIRY FARMS IN SOUTHERN AUSTRALIA

**CSIRO** PUBLISHING

*Animal Production Science*, 2014, **54**, 1960–1970
http://dx.doi.org/10.1071/AN14436

## Using a modelling approach to evaluate two options for improving animal nitrogen use efficiency and reducing nitrous oxide emissions on dairy farms in southern Australia

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**Abstract.** Ruminant livestock are generally considered inefficient converters of dietary nitrogen (N) into animal product. Animal nitrogen use efficiency (NUE) is a measure of the relative transformation of feed N into product and in dairy systems this is often expressed as milk N per unit of N intake (g milk N/100 g N intake). This study was a theoretical exercise to explore the relative potential efficacy and value proposition of breeding versus feeding to improve NUE, reduce urinary N excretion and associated environmental impact in pasture-based dairy systems. The biophysical whole farm systems model DairyMod was used across three dairying regions of south-eastern Australia representing a high-rainfall cool temperate climate (HRCT), a high-rainfall temperate climate (HRT) and a medium-rainfall temperate climate (MRT) to examine the two theoretical approaches of (1) maintaining the same amount of N exported in milk from a reduced N intake; and (2) increasing the amount of N exported in milk for the same amount of dietary N intake. Sixteen scenarios were explored for each site; these include four supplementary feed N (SN) concentrations (ranging from 1% to 4% N) combined with four milk N (MN) concentrations (ranging from 0.50% to 0.65% N). Reducing the SN concentration from 4% to 1% increased the 30-year mean model-predicted NUEs from ~16 g milk N/100 g N intake at all three sites to between 23 and 28 g milk N/100 g N intake, with the least and greatest improvements in NUE occurring for the HRCT and MRT sites, respectively. Corresponding to this improved NUE through reduced SN concentrations, model-predicted $N_2O$ emissions declined from 3.0 to 1.3 t carbon dioxide equivalents ($CO_2$-e)/ha.annum for the HRCT site, from 4.2 to 2.1 t $CO_2$-e/ha.annum for the HRT site and from 4.4 to 2.1 t $CO_2$-e/ha.annum for the MRT site, representing a decline of between 50% and 57%. In contrast, increasing the MN concentration from 0.50% to 0.65% increased the 30-year mean model-predicted NUEs from 17 to 22 g milk N/100 g N intake for the HRCT site, from 18 to 23 g milk N/100 g N intake for the HRT site and from 18 to 24 g milk N/100 g N intake for the MRT site. Corresponding to the improved NUE through increased MN concentrations, model-predicted $N_2O$ emissions declined from 2.3 to 2.0 t $CO_2$-e/ha.annum for the HRCT site, from 3.3 to 3.1 t $CO_2$-e/ha.annum for the HRT site and from 3.4 to 3.2 t $CO_2$-e/ha.annum for the MRT site; representing a decline of between 7% and 11%. These results suggest that improving animal NUE to reduce associated $N_2O$ losses holds much more promise if achieved through a reduction in the amount of N in supplementary feed than through increasing N exported in milk. This is an important finding for the Australian dairy industry, since manipulation of dietary N to better balance the energy to protein ratio would be much easier to implement than manipulation of N concentration in milk through genetics.

**Additional keywords:** dairy cows, DairyMod, milk protein, modelling, supplementary feeding, whole farm systems.

Received 24 March 2014, accepted 18 July 2014, published online 4 September 2014

## 5.1 Introduction

Ruminant livestock are generally considered limited in their ability to convert dietary N into animal product, with generally less than 30% of lifetime dietary N utilised for the production of meat, milk or fibre (Whitehead 1995). Given that the concentration of N in faeces and products remains relatively constant over a range of N intakes above minimum metabolic requirements, excess N is predominantly excreted in urine (Whitehead 1995). Any strategy that reduces the amount of excess N being consumed and excreted by livestock will generally have environmental benefits. Nitrous oxide ($N_2O$), a greenhouse gas (GHG) with 310 times the global warming potential of carbon dioxide ($CO_2$; DCCEE 2011d), is emitted to the environment through the process of denitrification, and to a lesser extent, nitrification within the soil (Dalal *et al.* 2003; de Klein and Eckard 2008). In addition, a proportion of N lost through the volatilisation of ammonia ($NH_3$) and the leaching/ runoff of nitrate ($NO_3$) will be re-deposited onto land by rainfall or through waterways and also result in indirect $N_2O$ emissions (DCCEE 2011d).

Animal N use efficiency (NUE) is a measure of the relative transformation of feed N into product. In dairy systems, animal NUE can be expressed as milk N per unit N intake (g milk N/100 g N intake). While many studies have explored the NUE of confinement feeding herds (Powell *et al.* 2010), fewer studies have been undertaken in the pasture-based grazing systems in the southern hemisphere where pasture forms a large component of the diet, with supplementary feed used to fill feed deficits at times of the year when pasture supply does not match animal demand (Vibart *et al.* 2009; Gourley *et al.* 2012a). Gourley *et al.* (2012a) found that measured NUEs varied between 15 and 35 g N milk/100 g N intake across 17 farms, where grazed pastures constituted the majority of the diet. They concluded that the highest NUE was generally obtained at the lowest level of feed N intake. In addition, numerous reviews have highlighted that improving NUEs is one pathway to reducing $N_2O$ emissions from livestock (de Klein and Eckard 2008; Eckard *et al.* 2010).

From a practical perspective, measuring N dynamics on farm is difficult, time consuming, expensive and can only consider a small combination of variables (e.g. soils, climate and management for a limited timeframe) (Bryant *et al.* 2011; Smith and Western 2013). Dynamic biophysical models provide an alternative for exploring these interactions over a longer timeframe and for varying farm management practices, climates and locations. The aim of the present study was to use a dynamic biophysical whole-farm system model to undertake a theoretical exercise to explore the relative potential efficacy and value proposition of breeding versus feeding to improve NUE, reduce urinary N losses and associated $N_2O$ emissions with variations in supplementary feed N (SN) concentrations and milk N (MN) concentrations for dairy farms in the temperate climates of south-eastern Australia.

## 5.2 Materials and methods

### *5.2.1. DairyMod*

The biophysical model DairyMod (version 4.8.16; Johnson *et al.* 2008) was used to simulate a 100 ha dairy farm, with animals grazing rain-fed perennial ryegrass (*Lolium perenne* L.) pasture sward in three south-eastern regions of Australia (Elliott, -41.1°N, 145.8°E; Ellinbank, -38.3°N, 146.0°E; and Terang, -38.3°N, 142.6°E). Daily weather data for each site were accessed as Patched Point Datasets from the Bureau of Meteorology SILO database (Jeffrey *et al.* 2001). The daily weather data used were rainfall (mm), minimum and maximum temperature (°C), minimum and maximum relative humidity (%), total solar radiation ($MJ/m^2$) and potential evapotranspiration (mm). Simulations were run between 1 July 1901 and 30 June 2000, with the financial years from 1970–1971 to 1999–2000 used for analysis. Over the simulation period, as an annual average, the daily 30-year mean temperature at Elliott was 15.7°C compared with 18.5°C and 18.6°C at Ellinbank and Terang, respectively. The 30-year mean (range) annual rainfall was 1220 (864–1860), 1083 (846–1339) and 787 (590–1006) mm at Elliott, Ellinbank and Terang, respectively. On the basis of these temperature and rainfall means, the three sites of Elliott, Ellinbank and Terang are representative of and herein referred to as a high-rainfall cool temperate (HRCT) site, a high-rainfall temperate (HRT) site and a medium-rainfall temperate (MRT) site, respectively.

DairyMod is a recognised whole farm systems-level computer-based model used for analyses of pasture based dairy systems in Australia and New Zealand (Cullen *et al.* 2008; White *et al.* 2008; Smith and Western 2013). Several authors have compared simulated pasture production estimates with measured data for the environments examined in this study, finding close agreement between the model-predicted and observed values (Chapman *et al.* 2008a; Cullen *et al.* 2008). Taking into consideration the many factors that can influence pasture growth rates (e.g. grazing management, N fertiliser inputs), and the difficulty of accurately measuring net herbage accumulation rate, it is not realistic to expect an exact match between predicted and observed growth rates (Chapman *et al.* 2008a). On the basis of the model evaluation statistics work of Tedeschi (2006), Cullen *et al.* (2008) found the *r2*, bias correction and mean prediction error to be 0.88, 1.0 and 10.8%, respectively, for 31 modelled and observed annual pasture yields across several temperate and subtropical environments, with varying ryegrass cultivars, soils types and management systems in Australian and New Zealand. Similar results have been found by other authors for some of the sites explored here, giving rise to confidence in DairyMod predicting dairy farm systems for south-eastern Australia.

DairyMod includes modules for pasture growth and subsequent utilisation by grazing animals, water and N dynamics, animal and plant physiology and production with a range of options for pasture management, irrigation and N fertiliser application. The animal module simulates animal production based on the metabolisable energy (ME) intake that is utilised for maintenance, lactation, pregnancy and growth as appropriate. The module has a significant role in predicting nutrient dynamics through the recycling of faeces and urine (herein referred to as excreta when summed together). The user defines the proportion of animal excreta that is returned to the paddock (see section 5.2.2.2 below). While DairyMod assumes an even distribution of excreta across paddocks, this is unlikely to be the case in reality. There is also the ability to define the nutrient composition of milk and this function was implemented to define the four MN concentrations examined in this study (see section 5.2.2.3 below). Information on how $N_2O$ emissions are estimated in DairyMod is provided in section 5.2.2.4 below. For more model details see Johnson (2013).

### *5.2.2. Farm system*

#### *5.2.2.1. Milking herd, grazing management and supplementary feed*

The same farm system was simulated at each site to avoid confounding comparison of results among sites. The farm system was a herd calving 1 August (late winter), with a target milk production of 6500 L/cow per 300-day lactation. This amount of milk production per cow was similar to regional mean values taken from the Victorian Department of Primary Industries' Dairy Industry Farm Monitor Project reports (Victorian Department of Primary Industries 2017) and Tasmanian Dairy Business of the Year award reports (Tasmanian Institute of Agriculture 2017). The N concentration of the milk was an experimental variable for this study and is detailed in section 5.2.2.3 below. The herd rotationally grazed 20 5-ha paddocks down to a biomass residual of 1.4 t dry matter (DM)/ha once the pasture reached the 2.5-leaf regrowth stage or if the pasture biomass had reached 2.5 t DM/ha, grazed once the pasture reach the 2.0-leaf stage of regrowth. The stocking rate was intentionally set high, at 3.5 cows/ha, to ensure a demand for supplementary feeding in all years. When daily pasture biomass was insufficient for daily herd ME-intake requirements, supplement with an ME concentration of 12.5 MJ/kg DM was supplied to meet the herd energy demand. Supplementary feed N concentration was the other experimental variable for this study and is detailed in section 5.2.2.3 below. DairyMod assumes 100% utilisation of all pastures grown and supplementary feed supplied to the herd (i.e. no spoilage due to trampling, excreta deposition on pastures etc).

#### *5.2.2.2. Soil parameters, N fertiliser and animal-waste management*

A generic clay loam soil with a bulk density value of 1.3 g/cm$^3$ and a saturated hydraulic conductivity of 2.8 mm/h was used for each site. The wilting point, field capacity and saturation point of the generic clay loam were 19%, 40%, and 48% by volume, respectively, with the perennial ryegrass pasture having a rooting depth of 400 mm. All sites were rain-fed, and given the winter-dominant rainfall pattern at all sites, N fertiliser was applied at a rate of 30 kg N/ha in the form of urea on the 1 April, 1 May, 1 August, 1 September and 1 October every year, to correspond with times of the year when response rate to the N fertiliser was most likely to be maximised. This N fertilisation rate was similar to regional benchmarking averages of ~120–130 kg N/ha.annum for Victorian dairy farms (Victorian Department of Primary Industries 2017) and ~170 kg N/ha.annum for Tasmanian dairy farms (Tasmanian Institute of Agriculture 2017).

In addition, 90% of animal excreta were returned to the paddocks, with the remaining 10% deposited in the dairy during milking. This reflects typical farm practices in Australia where the time cows spend grazing pastures is maximised and is equivalent to cows spending between 60 and 90 min per milking at the dairy and associated holding areas. To minimise any inconsistency in how the remaining 10% of animal excreta deposited at the dairy could be stored/managed between sites (e.g. flushed to a storage pond/lagoon system and spread at a later date or flushed to a sump and spread daily through a sprinkler system on pastures), and, consequently, alter the $N_2O$ emissions from this stored excreta, a consistent approach was taken, with these emissions effectively removed from the $N_2O$ estimations. Similar approaches have been undertaken by others to explore N dynamics using DairyMod (e.g. Smith and Western 2013).

#### *5.2.2.3. Milk N concentrations and supplementary feed N concentrations*

The following sixteen scenarios were examined for each site: four MN concentrations of 0.50%, 0.55%, 0.60% and 0.65% N, representing 3.1%, 3.4%, 3.8% and 4.1% milk protein, combined with four SN concentrations of 1%, 2%, 3% and 4% N, representing 6.3%, 12.5%, 18.8% and 25.0% crude protein in the supplement. Milk concentrations of either 0.50% or 0.55% N are typical of Holstein-Friesian cattle where increased milk volume generally results in a dilution of milk protein concentrations (Robinson 2014; http://animalscience.ucdavis.edu/faculty/robinson/Articles/FullText/Pdf/Web199908.

PDF, accessed 26 August 2014). In contrast, milk concentrations of 0.60% N are typical of Jersey cattle, and to a lesser extent Guernseys, Ayrshires and cross-bred cattle, where there is generally a decrease in milk volume, resulting in a higher concentration of milk protein (Robinson 2014). The fourth 0.65% MN was included to examine an extreme level of MN concentration. Long-term increases in MN concentrations are generally possible only through changes in cattle breed, and to a lesser extent genetic selection within breeds, with changes in feeding regimes resulting only in transitory changes in MN concentrations (Robinson 2014). The SN concentrations of either 3% or 4% N in the current study are reflective of an all pasture diet; replacing grazed pastures with supplementary feed of a similar N concentration and therefore essentially not varying the overall diet N concentration by any substantial amount. In comparison, the SN concentrations of 2% N reflects a pasture based diet supplemented with grain-based concentrates, while the 1%SN reflects a pasture-based diet supplemented with forage of a low N concentration, such as maize (*Zea mays* L.) silage.

### *5.2.2.4. Nitrous oxide emissions*

Denitrification losses are from the soil $NO_3$ pool, which is supplied either through direct application of $NO_3$ fertiliser or from the nitrification of ammonium ($NH_4$). Ammonium inputs are from organic matter breakdown (including dung decay), inputs from $NH_4$ fertiliser and urine inputs. Nitrogen in the diet is either retained in body tissue growth, converted into milk or excreted. It is assumed that the milk has a fixed N concentration throughout the simulations and we altered this fixed concentration to create the four MN concentrations. Partitioning of N between dung and urine is related to the N concentration of the diet and proportion of N that is excreted. The general behaviour of DairyMod is that the proportion of excreted N in urine increases as the N content of the diet increases (Johnson *et al.* 2008).

Nitrification of $NH_4$ is defined using a rectangular hyperbola in response to $NH_4$ concentration, and is also affected by soil water status, temperature and soil microbial pools (Johnson *et al.* 2008). The rate of denitrification is defined using a rectangular hyperbola in response to soil $NO_3$ concentration, as well as temperature and soil C status, with soil C status varying through the depth of the soil profile as defined by the ratio of labile soil C in each soil layer to that in the surface layer (Johnson *et al.* 2008). Partitioning of denitrification losses between $N_2O$ and $N_2$ is affected by water-filled pore space (WFPS) based on the model proposed by Granli and Bøckman (1994). Denitrification of $N_2O$ commenced and ceased at a WFPS of 60% and 90%, respectively. Denitrification of $N_2$ commenced at a WFPS of 80%.

Indirect $N_2O$ losses, which are the proportion of leached $NO_3$ and volatilisation of $NH_3$ being converted into $N_2O$ at a later date, were estimated using an inventory approach. Several studies have found consistency with the model-prediction of N leaching, volatilisation and/or denitrification within DairyMod compared with measured data (Bryant *et al.* 2011; Hoogendoorn *et al.* 2011; Smith and Western 2013) giving the model credibility in estimating N dynamics for pasture-based grazing systems. Simulated losses of N through leaching and volatilisation were multiplied by emissions factors of 0.01 and 0.0125, respectively, as used in the Australian National GHG Inventory (DCCEE 2011d). A global warming potential of 310 was used to convert $N_2O$ losses to GHG carbon dioxide equivalent emissions (kg $CO_2$e; DCCEE 2011d). For more details on DairyMod, see Johnson (2013).

### *5.2.3. Model outputs*

Model-predicted annual data for pasture intake (t DM/ha), supplementary feed intake (t DM/ha), milk production (L/ha), N fertiliser applied (kg N/ha), N exported in milk (kg N/ha), animal N intake (kg N/ha), animal excreta returned to paddocks (kg N/ha), direct $N_2O$ emissions (kg $CO_2$e/ha), leached N (kg N/ha) and volatilised N (kg N/ha) were collated for each SN by MN scenario and site over the 30-year period to estimate the NUEs and $N_2O$ emissions.

### *5.2.4. NUE and $N_2O$ emissions estimations*

Annual NUE was determined for each of the 16 scenarios by dividing the annual total N exported in milk by the annual total N consumed by the cows (g milk N/100 g N intake) and from this, a 30-year mean annual NUE was determined for each scenario and site. Thirty-year mean NUEs were estimated for each of the four SN concentrations by averaging the mean NUEs for the four MNs within each SN concentration (e.g. average of the 30-year mean NUEs of the four MNs where SN was e.g. 1%, 2%). The process was repeated for the four MN concentrations by averaging the mean NUEs for the four SNs within each MN concentration (e.g. average of the 30-year mean NUEs of the four SNs where MN was e.g. 0.50%, 0.55%). In addition, 30-year mean $N_2O$ emissions (directly as $N_2O$ and indirectly through leaching and volatilisation) for each of the four SN concentrations (average of the four MNs) and each of the four MN concentrations (average of the four SNs) were estimated for each site using the same process as above.

### *5.2.5. Statistical analyses*

Thirty-year means and standard deviations (SDs) were estimated for annual pasture production and supplementary feed intakes for each site across all 16 scenarios. In addition, 30-year means and SDs were estimated for each of the four mean SN and MN concentration scenarios with respect to N inputs (dietary N intake) and outputs (N deposited on pastures from excreta, N exported in milk, N lost through denitrification, leaching, volatilisation and total $N_2O$ losses) for each site.

## 5.3 Results

### *5.3.1. Pasture production and supplementary feed intake, diet N intakes, milk production and animal excreta*

The simulated 30-year mean (± s.d.) annual pasture production was 11.6 (± 1.7), 9.4 (± 1.9) and 8.3 (± 1.6) t DM/ha for the HRCT, HRT and MRT sites, respectively. These mean annual pasture yields align with previous model validation studies at these sites (Cullen *et al.* 2008). On a per cow basis, the model predicted 30-year mean pasture intake equated to 3.3 (± 0.5), 2.7 (± 0.5) and 2.4 (± 0.5) t DM/cow.annum for the HRCT, HRT and MRT sites, respectively (Figure 5.1). Due to the inter-annual variation in pasture production, the model-predicted 30-year mean (± s.d.) supplementary feed intakes were 2.2 (± 1.6), 2.8 (± 1.8) and 3.1 (± 1.6) t DM/cow.annum for the HRCT, HRT and MRT sites, respectively (Figure 5.1). The amount of supplementary feeding was greater than industry averages. However, this was due to the high stocking rate, which was specifically selected such that there was a requirement for supplementary feed even for high pasture-production years. The percentage of total diet from supplementary feed, as a mean (± s.d.) of all 16 scenarios across all 30 years, were 40 (± 8), 50 (± 10) and 57% (± 8) for the HRCT, HRT and MRT sites, respectively.

Total diet N concentration varied between 3.1% and 3.3% N among the four model-predicted 30-year mean MN concentrations. When fed a supplement with an N concentration of 1%, mean (± s.d.) dietary N concentrations were 2.5 (± 0.3), 2.4 (± 0.2) and 2.2 (± 0.3)% for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4). This increased to 4.1% N across all sites when the SN concentration was increased to 4% N and was comparative to that of the pasture N concentrations (Figures 5.2 to 5.4). Not dismissing the complexity of microbial N dynamics within the cow, optimum concentrations of N in the diet for high-yielding dairy cows fed a well-balanced diet is generally accepted within the range of 2.6–2.9% (Olmos Colmenero and Broderick 2006), making the 1% SN diet marginal at all three sites.

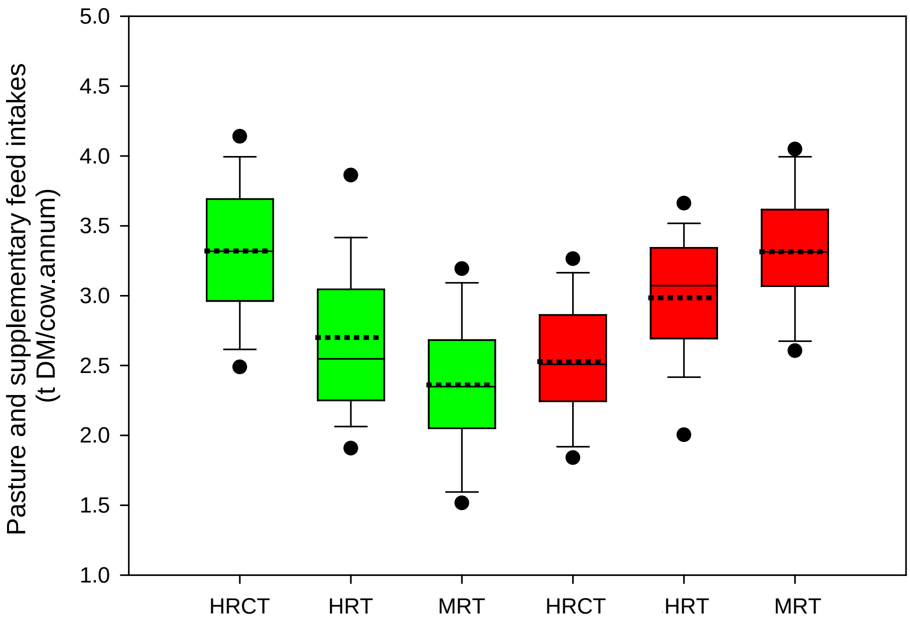


**Figure 5.1** Boxplot showing the estimated annual pasture intakes (green) and supplementary feed intakes (red) for a high-rainfall cool temperate climate (HRCT), a high-rainfall temperate climate (HRT) and a medium-rainfall temperate climate (MRT) between 1971 and 2000. (Boxplots represent 25th and 75th percentiles, whiskers represent 10th and 90th percentiles, outliers represent 5th and 95th percentiles, solid lines represent medians and dotted lines represent means).

### *5.3.2. Diet N intakes, milk production and animal excreta*

The model-predicted 30-year mean (± s.d.) annual N intakes were 539 (± 48), 483 (± 52) and 456 (± 49) kg N/ha for the HRCT, HRT and MRT sites, respectively, when the SN was 1% (Figures 5.2 to 5.4). For every 1% increase in SN concentrations, the 30-year mean total diet N intakes increased by ~90, 100 and 110 kg N/ha.annum for the HRCT, HRT and MRT sites, respectively. In addition, the inter-annual variation in dietary N intakes declined considerably as SN increased to 4% (Figures 5.2 to 5.4). In contrast, there was very little (<5 kg N/ha.annum) variation in the total diet N intakes across the four MN scenarios (Figures 5.2 to 5.4).

The overall 30-year model-predicted mean milk production across all 16 scenarios was ~6190 L/cow.lactation at all three sites, close to the initial target of 6500 L/cow.lactation, with little inter-annual variation within and between sites. This resulted in similar amounts of N exported in milk each year, with 30-year model-predicted means of 108, 119, 130 and 140 kg N/ha.annum (s.d. <1 kg N/ha.annum at all three sites) when the MN concentrations were 0.50%, 0.55%, 0.60% and 0.65%, respectively. For every 0.01% increase in MN concentrations, the model-predicted 30-year mean milk N exports (averaged over the four SN scenarios) increased by ~2.2 kg N/ha.annum.

As SN concentrations increased from 1% to 4%, so too did the amount of N returned to the paddock in excreta. There were substantial differences in N excreta among sites when the SN concentration was 1% N, as shown with a model-predicted 30-year means (± s.d.) of 359 (± 45), 309 (± 49) and 285 (± 45) kg N/ha for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4). These results align with measured studies where intakes similar to those achieved with the SN 1% diets reported above resulted in between ~250 and 315 g N excreted/cow.day (Kebreab *et al.* 2001). However, once the SN concentration was either 3% or 4% N, there was less variation among sites in the amount of N returned to the paddock in excreta, best illustrated with the 4% SN, with means (± s.d.) of 590 (± 13), 576 (± 13) and 580 (± 12) kg N/ha for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4). Increasing the MN concentration from 0.50% to 0.65% resulted in a reduction in N lost via excreta. However, this decline was relatively insignificant (~10 kg N/ha.annum per 0.05% MN increase) compared with the rate of decline in excreta N with declining SN concentrations (~70–100 kg N/ha.annum per 1% SN decline).

### *5.3.3. Nitrogen use efficiency*

As SN concentrations increased, mean NUE decreased. In contrast, NUE increased with increasing MN concentrations (Figures 5.2 to 5.4). The 30-year mean NUEs varied between 14 and 31 g milk N/100 g N intake (Figures 5.2 to 5.4). There was little difference in NUEs between sites when the SN concentration was 3% or 4% (Figures 5.2 to 5.4). However, when the SN concentration declined to 1%, NUE differences became more apparent, with mean NUEs (± s.d.) of 23 (± 3), 26 (± 4) and 28 (± 4) g milk N/100 g N intake for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4).

### *5.3.4. N lost through denitrification, leaching and volatilisation*

When the SN was 1%, mean (± s.d.) N lost to the environment directly through $N_2O$ denitrification was 1.1 (± 0.4), 2.3 (± 0.8) and 2.4 (± 0.6) kg N/ha for the HRCT, HRT and MRT sites, respectively. As SN increased from 1% to 4% N, the amount of N lost as $N_2O$ increased at all three sites, but with a diminishing rate of increase. For example, for the MRT site, the mean $N_2O$ model-predicted to be lost to the environment was 2.4, 2.9, 3.3 and 3.5 kg N/ha.annum for SNs of 1%, 2%, 3% and 4%, respectively. As MN increased, there was only a small reduction in the amount of N lost as $N_2O$, with a maximum difference of only 0.2 kg N/ha.annum between and MN of 0.50% and 0.65%.

Nitrogen was also lost to the environment through leaching and, to a lesser extent, volatilisation. Given that the simulated volatilisation losses were only ~10% of those losses from leaching, only the leaching results are presented here. However, the amount of $N_2O$ attributed to volatilisation losses is included in the total $N_2O$ losses presented in section 5.3.5 below. Similarly to N lost as $N_2O$ through denitrification, the amount of N lost through leaching was the lowest for the HRCT site and greatest at the MRT site. When SNs were 1%, the 30-year mean (± s.d.) leached N was 109 (± 52), 147 (± 61) and 148 (± 63) kg N/ha.annum for the HRCT, HRT and MRT sites, respectively. As SN increased from 1% to 4%, the model-predicted 30-year mean leached N increased by 181, 227 and 257 kg N/ha for the HRCT, HRT and MRT sites, respectively. In contrast, as MN increased from 0.50% to 0.65%, the 30-year mean leached N declined by between 24 and 25 kg N/ha across the three sites.

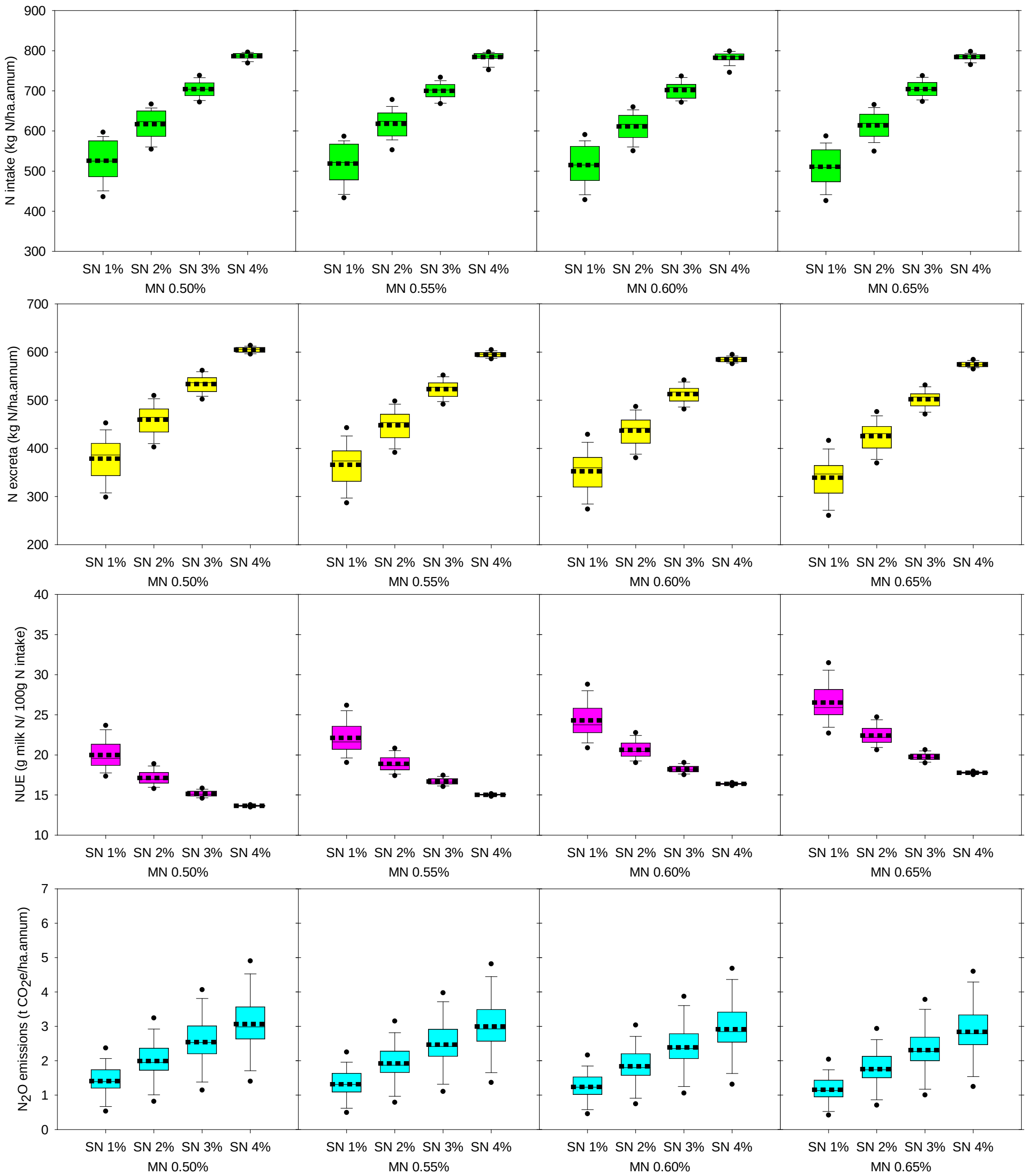


**Figure 5.2** Boxplots showing the estimated nitrogen intake (kg N/ha.annum; ), nitrogen excreted in dung and urine (kg N/ha.annum; ), nitrogen use efficiency (g milk N/ 100g N intake; ) and nitrous oxide emissions (t $CO_2$e/ha.annum; ) for the high-rainfall cool temperate climate site between 1971 and 2000 with milk nitrogen (MN) concentrations of 0.50, 0.55, 0.60 and 0.65% and supplementary feed nitrogen (SN) concentrations of 1, 2, 3 and 4%. (See Figure 5.1 for boxplot interpretation).

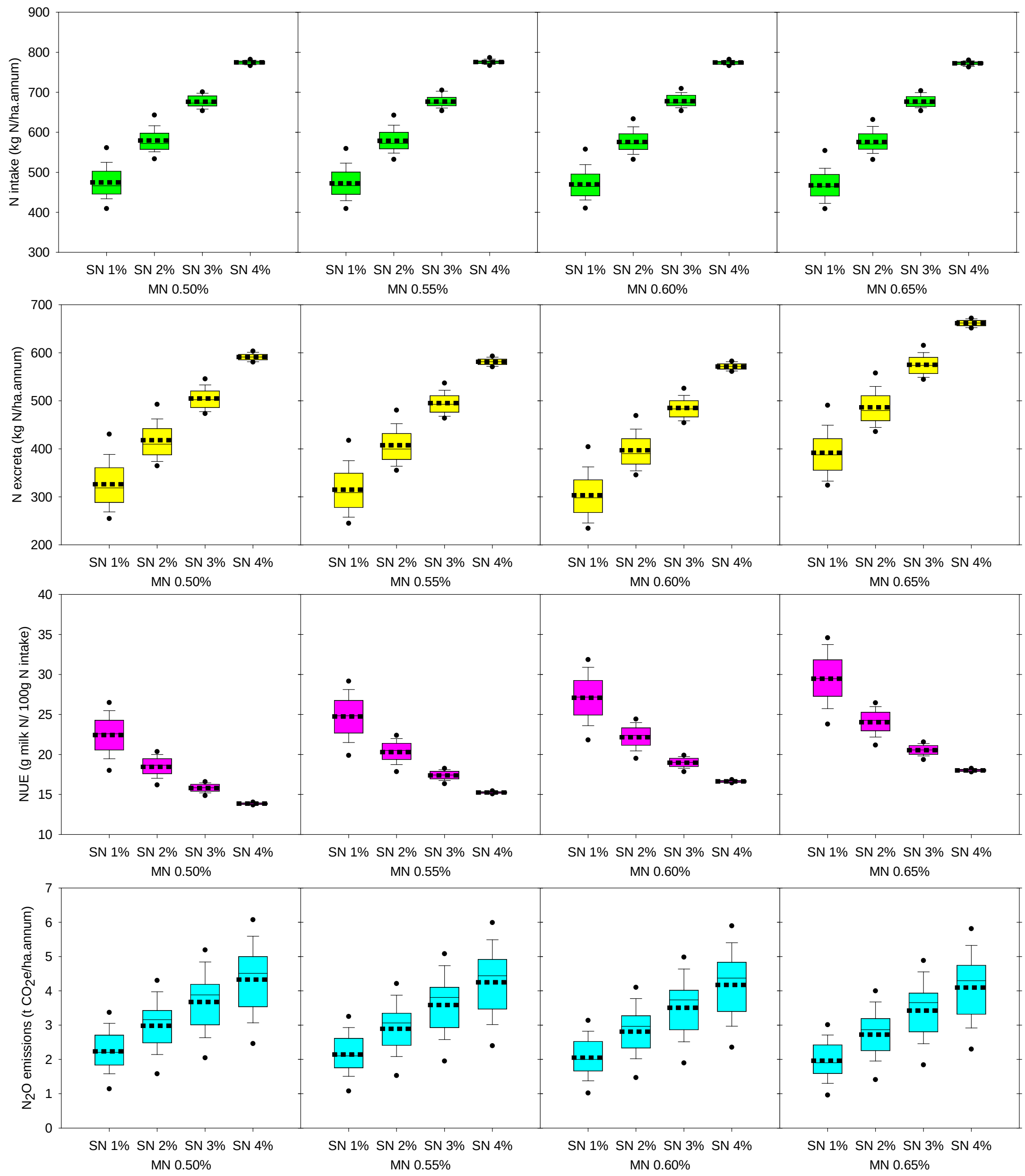


**Figure 5.3** Boxplots showing the estimated nitrogen intake (kg N/ha.annum; ), nitrogen excreted in dung and urine (kg N/ha.annum; ), nitrogen use efficiency (g milk N/ 100g N intake; ) and nitrous oxide emissions (t $CO_2$e/ha.annum; ) for the high-rainfall temperate climate site between 1971 and 2000 with milk nitrogen (MN) concentrations of 0.50, 0.55, 0.60 and 0.65% and supplementary feed nitrogen (SN) concentrations of 1, 2, 3 and 4%. (See Figure 5.1 for boxplot interpretation).

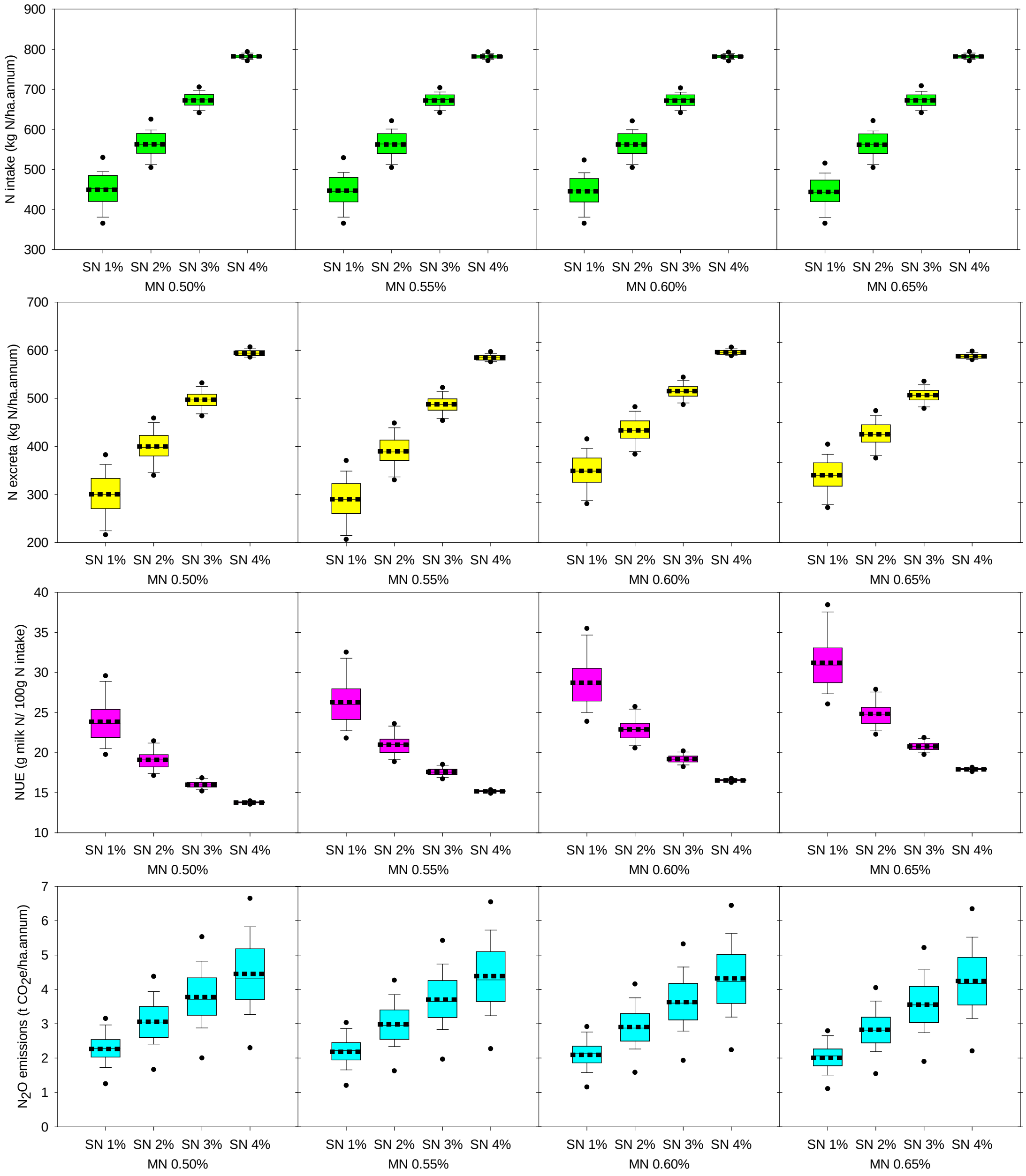


**Figure 5.4** Boxplots showing the estimated nitrogen intake (kg N/ha.annum; ), nitrogen excreted in dung and urine (kg N/ha.annum; ), nitrogen use efficiency (g milk N/ 100g N intake; ) and nitrous oxide emissions (t $CO_2$e/ha.annum; ) for the medium-rainfall temperate climate site between 1971 and 2000 with milk nitrogen (MN) concentrations of 0.50, 0.55, 0.60 and 0.65% and supplementary feed nitrogen (SN) concentrations of 1, 2, 3 and 4%. (See Figure 5.1 for boxplot interpretation).

#### *5.3.5. Total nitrous oxide emissions*

Total $N_2O$ emissions, as the sum of the $N_2O$ lost directly and through the conversion of a proportion of N lost through leaching and volatilisation into $N_2O$, varied between scenarios and sites. When the SN was 4% N, the model-predicted 30-year mean (± s.d.) total $N_2O$ emission was 3.0 (± 0.9), 4.2 (± 1.0) and 4.4 (± 1.1) t $CO_2$e/ha.annum for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4). This declined to model-predicted 30-year mean (± s.d.) total $N_2O$ emissions of 1.3 (± 0.5), 2.1 (± 0.6) and 2.1 (± 0.4) t $CO_2$e/ha.annum for the HRCT, HRT and MRT sites, respectively, when SN was 1% N (Figures 5.2 to 5.4). When the MN concentration was 0.50% N, the model- predicted 30 year mean (± s.d.) total $N_2O$ emission was 2.3 (± 0.9), 3.3 (± 1.1) and 3.4 (± 1.1) t $CO_2$e/ha.annum for the HRCT, HRT and MRT sites, respectively (Figures 5.2 to 5.4). This declined to 2.0 (± 0.9), 3.1 (± 1.1) and 3.2 (± 1.1) t $CO_2$e/ha.annum when the MN concentration was increased to 0.65% N (Figures 5.2 to 5.4).

Direct $N_2O$ emissions, as a mean of the 16 scenarios, were model-predicted as 0.39, 0.48 and 0.48 of total $N_2O$ emissions for the HRCT, HRT and MRT sites, respectively, with the balance attributed to indirect $N_2O$ emissions associated with leaching and volatilisation. These proportions are similar to those of the national inventory (DCCEE 2011d), giving additional weight to the accuracy of the model outputs.

## 5.4 Discussion

On the basis of the assumptions inherent in DairyMod, the present study suggests that decreasing dietary N supply to better balance the dietary N requirements of a dairy cow leads to improvements in NUE and results in greater reduction in $N_2O$ emissions than strategies that are targeting a greater amount N in the animal product. In practice, improving the balance between protein and energy can be achieved by several processes. The most obvious is to provide a diet that matches N requirement, varying throughout the year depending on animal requirements for lactation and pregnancy. However, for pasture-based systems typical of southern Australia, where there can be large seasonal variations in pasture N concentrations, this is somewhat more difficult to achieve (Powell *et al.* 2010). Options for pasture-based dairy systems include selecting forage species with a higher energy to protein ratio, strategically feeding animals on the basis of changes in N concentration with plant phenology or providing supplements with a low N concentration (de Klein and Eckard 2008). Such feeding strategies can be more easily adapted in Australia's pasture-based dairy systems than longer term options such as selection of dairy cows for a higher concentration of milk protein (Keim and Anrique 2011), especially given that the assumed theoretical increases in milk N concentration presented here resulted in minimal estimated potential reductions in $N_2O$ emissions.

In the current study, the 30-year mean NUEs for individual scenario were model-predicted to vary between 14 and 31 g milk N/100 g N intake. These results are similar to measured results of Gourley *et al.* (2012a) who found that measured NUEs of 17 dairy farms located throughout Victoria, including some farms located in the HRT and MRT regions explored in the present study, varied between 15 and 35 g milk N/100 g N intake. In a review of NUEs by Keim and Anrique (2011), where pastures comprised the majority of the diet, measured NUEs varied between 15 and 33 g milk N/100 g N intake. Similar measured results were found by Vibart *et al.* (2009), with an average NUE of 24 g milk N/100 g N intake.

According to the definitions of varying NUEs by Chase (2003), NUEs of < 25 g milk N/100 g N intake are considered to indicate room for substantial improvement. However, these definitions by Chase (2003) were based on studies in which most cattle were involved in confinement feeding practices, as opposed to the majority of the diet consisting of grazed pastures for dairy farms in southern Australia (Dairy Australia; http://www.dairyaustralia.com.au/~/media/Documents/Animal%20 management/Feed%20and%20nutrition/Feeding%20Systems %20latest/Aus%20five %20main%20feeding%20systems.pdf, accessed 26 August 2014). Compared with grazed pastures, confinement feeding allows for more control of feed quantity and nutrient concentrations throughout the year, where the energy to protein ratio can be monitored and rectified as required, the diet can be delivered in smaller amounts throughout the day to increase daily dry matter intakes and milk production per unit of feed intake can be maximised due to less energy being diverted to maintenance and activity (Powell *et al.* 2010; Gourley *et al.* 2012a). Any one of these would assist in increasing the NUE of milk production for confinement feeding systems relative to grazed-pastures systems typical of southern Australia.

When the SN concentration declined from 4% to 1%, reflecting a change in overall diet N concentration from 4.1% down to a low of 2.2%, depending on the site, N intakes also declined from ~790 kg N/ha.annum at all three sites to between 460 and 540 kg N/ha.annum, depending on the site. This corresponded with an improvement in NUE from a low of 16 g milk N/100 g N at all three sites to a high of between 23 and 28 g milk N/100 g N. This relationship between N intake and NUEs has been shown in measured studies, highlighting that any practice that better balances N intake will improve NUE (Huhtanen and Hristov 2009; Powell *et al.* 2010).

When MN concentrations increased from 0.50% to 0.65%, N intakes remained relatively stable and NUEs increased from ~17 to 23 g milk N/100 g N. More importantly, this increase in NUE was lower than that achieved with reducing SNs. Milk N concentrations of 0.50% N returned a 30 year mean of between 71% and 73% of total N intake as excreta and only declined to between 66% and 68% of total N intake as excreta when the MN concentration was increased to 0.65% N. Keim and Anrique (2011) suggested that increasing the amount of N exported in milk is a positive means of improving NUE to reduce $N_2O$ losses. However, the reduction in $N_2O$ corresponding with improved NUEs, as a consequence of increased MN concentrations, resulted in 0.2 – 0.3 t $CO_2$e/ha.annum abatement, depending on the site. This was a relatively insignificant abatement compared with those achieved with declining SN concentrations.

Due to climatic differences, pasture consumption for the HRCT site was 11.6 t DM/ha.annum, which was discernibly higher than the 9.4 and 8.3 t DM/ha.annum achieved for the HRT and MRT sites, respectively. This affected the N balance in two ways, namely, more of the N inputs from fertiliser and excreta were converted into pasture and less supplementary feed was required. Both aspects would have contributed to less N being available for loss, either directly or indirectly, highlighting that maximising pasture production and pasture utilisation is the goal of dairy farmers to not only better balance their herd's feed requirements, but also reduce some of the environmental concerns of dairy farming.

Direct and indirect $N_2O$ emissions for the HRCT site were about half and one-third those at the other two sites further confirming that converting N into pasture production attributed to the reduced $N_2O$ losses for the HRCT site. The warmer soil temperatures experienced at the two temperate climate sites may have also favoured increases in $N_2O$ emissions relative to those at the cool temperate climate site, even though the latter was discernibly wetter over the winter months (Dalal *et al.* 2003).

Therefore, for the temperate climate sites, the implementation of other mitigation strategies in addition to better balancing of the diet energy to protein ratio could have some merit. Other potential strategies to optimise pasture production in addition to reducing $N_2O$ emissions include management practices such as improved drainage, better scheduling of irrigation events to minimise through drainage and/or strategic grazing of pastures over winter to reduce soil conditions conducive to N losses (de Klein and Eckard 2008).

For many southern Australian dairy systems, diets are predominantly grazed and conserved pastures based on perennial ryegrass (Chapman *et al.* 2008a). This study has shown that the introduction of some form of forage crop low in N concentration, such as maize grown for silage, was advantageous in reducing $N_2O$ emissions. However, although the use of forage crops to better balance the diet can be beneficial, it is critical to avoid inadvertently introducing an energy deficit as some of these low N forages can also be below optimum energy (e.g. sorghum (*Sorghum bicolour* L) and millet (*Pennisetum glaucum* L)), leading to a compromise in milk production and, subsequently, increasing the NUEs. In addition, on-farm adoption of forage crops is often low, as improvements in farm profitability can be minimal (Chapman *et al.* 2008b; Rawnsley *et al.* 2013). In addition, their introduction could bring changes to management practices for the farming system. For example, the inclusion of maize silage into the diet may require specialised harvesting equipment, concreted feeding areas and mixer wagons for feeding. Therefore just replacing a proportion of the diet with a low N concentration forage supplement has other considerations, which need to be addressed beyond their ability to reduce $N_2O$ losses.

The present study has clearly demonstrated, and supports previous findings, that adopting feeding strategies to better balance the diet so that N is not supplied in excess of animal requirements will most likely result in improved NUE and, consequently, lower $N_2O$ emissions (Powell *et al.* 2010; Keim and Anrique 2011). Identifying pasture/forage species that can reduce N intakes without compromising milk production and farm profitability will have positive outcomes for the environment. In recent years, there has been an increasing focus on the use of high-sugar ryegrass to improve the energy to protein ratio of the cow diet (Keim and Anrique 2011). The study has shown that any strategy that reduces the amount of surplus N in the diet will result in improvements in NUE and, consequently, decreases $N_2O$ emissions. This adoption of high-sugar ryegrass in temperate dairying regions of Australia requires further research to ascertain both the productive and environmental benefits.

### 5.5 Conclusions

It is generally accepted that while there are several pathways to improve animal NUE for confinement feeding of dairy cows, large uncertainty remains regarding pathways for improving NUEs in pasture-based dairy systems (Keim and Anrique 2011; Gourley *et al.* 2012). Finding approaches to achieving improvements in NUE to reduce associated $N_2O$ emissions is considered critical to maintaining productivity growth whilst adhering to environmental stewardship. The present study explored two potential pathways for improving NUE to assist in reducing $N_2O$ emissions, namely, increasing the amount of N exported off-farm with improved milk N concentration and decreasing the overall N concentration of the diet with varying supplementary-feed N concentrations. Both pathways improved NUE and reduced associated $N_2O$ emissions. However, reducing supplementary feed N concentration reduced N losses by an order of magnitude greater than achieved by increasing milk N concentrations. It is also clear that no single strategy alone will achieve the desired improvement in NUE and $N_2O$ abatement; however, a combination of several complementary options may hold some promise for the southern Australian dairy industry.

### 5.6 Acknowledgements

This project was supported by the Tasmanian Institute of Agriculture, the University of Melbourne and the Victorian Department of Environment and Primary Industries, through funding from the Australian Government Department of Agriculture, Dairy Australia, Meat & Livestock Australia and Australian Wool Innovation.

# CHAPTER 6 MODELLING ENTERIC METHANE ABATEMENT FROM EARLIER MATING OF DAIRY HEIFERS IN SUBTROPICAL AUSTRALIA BY IMPROVING DIET QUALITY

CSIRO PUBLISHING

*Animal Production Science*, 2016, **56**, 565–573
http://dx.doi.org/10.1071/AN15296

## Modelling enteric methane abatement from earlier mating of dairy heifers in subtropical Australia by improving diet quality

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**Abstract.** Milking cows typically dominate dairy farm greenhouse gas (GHG) emissions, but replacement heifers also contribute to farm emissions and can increase the emission intensity of milk production. In northern Australia, heifers generally graze poorer-quality subtropical pastures and in the absence of energy-dense supplementary feed during periods of low pasture growth, liveweight (LW) gain can be restricted. This modelling study examined the time required and enteric methane ($CH_4$) emissions produced in raising dairy heifers to a target LW for first mating by feeding a diet assuming either constant (static) or variable (dynamic) nutritive values. Using a static approach (Australian Feeding Standards methodology), and assuming a target mating LW of 360 kg, growing heifers reached their target LW at ~18 months of age while consuming $C_4$ grasses with a constant metabolisable energy content of 9.5 MJ/kg dry matter (DM) or 11 months of age on a diet of 11.0 MJ/kg DM. Enteric $CH_4$ emissions were 1.2 and 0.8 t of carbon dioxide equivalents/heifer over the 18- and 11-month periods, respectively. To explore the extent with which climatic conditions influence seasonal pasture availability and nutritive value with a dynamic approach, we used a whole-farm biophysical model (SGS pasture model) to simulate diets with mean metabolisable energy values of 9.5 and 10.9 MJ/ kg DM. On average (±s.d.), heifers required 22 ± 4 and 17 ± 1 months, respectively, to reach target LW, with cumulative enteric $CH_4$ emissions of 1.22 ± 0.20 and 0.72 ± 0.04 t carbon dioxide equivalents, respectively. The dynamic approach resulted in slower LW gain due to the variable nutritive value of the diet throughout the year, resulting in seasonal periods of LW plateauing or decline. Maintaining heifers on high-quality diets in subtropical northern Australia should result in increased daily LW gain, lower enteric $CH_4$ emissions to mating LW and earlier calving. Together, these factors reduce their lifetime emission intensity of milk production.

**Additional keywords:** greenhouse gas emissions, liveweight gain, SGS pasture model, subtropical pastures.

Received 11 June 2015, accepted 28 October 2015, published online 9 February 2016

## 6.1 Introduction

Greenhouse gas (GHG) emissions from agriculture contribute ~14.5% of global emissions (Gerber *et al.* 2013a) and improvements in farm system management can help abate such emissions. While the milking cow is the single largest source of on-farm dairy GHG emissions, replacement heifers also contribute and more importantly emit GHG without producing milk, thus increasing the GHG emission intensity of milk production from the whole farm. The average age of first calving on Australian dairy farms varies considerably and generally reflects whether the region supplies liquid milk or manufactured products destined for export. Traditionally, there has been a trend towards a flatter annual milk production curve in northern Australia to supply the liquid milk market, resulting in a dairy farm that accommodates heifers calving at an older age and at any time during the year (e.g. Kempton and Waterman 2014). This is best illustrated by an average calving age of ~33 months for the northern Australian dairy industry, reflecting a first mating age of 24 months (Hough 1992). In contrast, in southern Australia, there are generally two dominant peak periods of milk supply, in spring and, to a lesser extent, autumn, to reflect seasonal climatic conditions and associated pasture growth (e.g. Gilmour *et al.* 2012). Heifers are traditionally mated to coincide calving with these more opportunistic seasonal periods as shown with a first-calving average age of 27 months in Tasmania and Victoria, reflecting a mating age of 18 months (Hough 1992).

A preliminary survey of herd recording data collected between 1992 and present for the northern dairy industry suggests that the average age to first calving has risen slightly to 34 months in recent times (L. M. Trevaskis, unpubl. data). The slow growth of heifers in the northern dairying regions of Australia is generally associated with the use of tropical C4 pasture species, which are typically higher in fibre and lower in dry matter (DM) digestibility and crude protein than temperate C3 pasture species, especially if managed suboptimally (Moss 1993; Fulkerson *et al.* 1998). Heifer growth rates have been reported to be as low as 0.25 kg/day when grazing low energy-density tropical pastures without supplementation (Moss 1993).

Dobos *et al.* (2001) found that, assuming heifers calved at a similar liveweight (LW), the first-lactation milk production of animals calving at 2 years of age was 88% of those calving for the first time at 3 years of age. By the end of their third lactation, differences in milk production per lactation were shown to be minimal between the two calving-age groups (Cowan *et al.* 1974; Dobos *et al.* 2004), implying that early mating will increase lifetime milk production. Reducing the mating age has other benefits, such as reducing the number of replacement heifers required to maintain a similar milking-herd size, and acceleration in genetic improvement of the herd (Hoffman and Funk 1992; Moss 1993).

The Australian dairy industry has set a target of reducing the industry GHG emission intensity by 30% by the Year 2020, relative to 2010 emission intensity (Australian Dairy Industry Council 2013). This will be achieved only if several mitigation options are explored and implemented simultaneously across the whole farm system. Mating older heifers increases GHG emissions per unit product produced over the animal's lifetime, since these animals are emitting methane ($CH_4$) from enteric fermentation as well as $CH_4$ and nitrous oxide ($N_2O$) losses from dung and urine deposition, without generating any milk for a longer period of their lifetime.

In the current study, we hypothesised that the latitudinal range in age at first mating spanning the north–south dairy regions of Australia reflects differences in total diet energy availability, especially the quality of grazed pasture. The most obvious difference is the higher proportion of tropical C4 grasses that make up the diet of heifers in northern Australia. To explore this hypothesis, we estimated the relative time to reach a mating LW of 360 kg and the associated enteric $CH_4$ emissions when maiden heifers were fed a range of diets with varying nutritive value reflective of local conditions. We examined animal growth rates and GHG emissions using an approach that assumed that daily energy supply and animal growth rates were constant (static approach) and compared this with a dynamic approach conducted using a biophysical model that accounted for seasonal variation in pasture nutritional quality and heifer liveweight gain (LWG) over time.

## 6.2 Materials and methods

Two forms of desktop analysis were undertaken in the current study. In the static approach, we assumed that feed nutritional characteristics were constant over the course of the year. In the dynamic approach, feed nutritional characteristics varied throughout the year depending on seasonal climatic conditions, pasture availability and their compounding effect on LWG. Both analyses were based on data obtained from case-study farms where heifers grazed kikuyu (*Pennisetum clandestinum*) pastures (see Supplementary Material in section 6.5).

### *6.2.1. Static analysis*

The static analysis was conducted to determine the average daily LWG, cumulative $CH_4$ emissions and associated time between weaning of dairy heifers, at 100 kg LW and 100 days old, and mating at 360 kg. This approach was computed using the Feeding Standards for Australian Livestock – Ruminants (CSIRO 2007). Heifers were offered a diet containing 100% C4 grasses with varying nutritive values of 9.5, 10.0, 10.5 or 11.0 MJ metabolisable energy (ME)/kg DM, reflecting values typical of or at the upper range for the study region (Fulkerson *et al.* 1998, 2007; Garcia *et al.* 2008, 2014). DM intake (DMI, kg DM/day. head) was calculated as a function of LW, mature body size (assumed at 600 kg throughout the present study), relative size (ratio of daily LW to mature LW) and nutritive value. Total ME intake (MEI) was calculated as the product of intake and ME density of the feed (MJ ME/kg DM), summed over time. The ME required for grazing uses several constants, DM intake, LW and an estimation of horizontal equivalent of distance walked on the basis of stocking rate (assumed to be 3.5 heifers/ha on the basis of farm data), average pasture biomass (assumed to be an average of 1.5 t DM/ha over the duration of the study), distance walked (assumed to average of 1.0 km/day over the study duration) and slope (scale from 1 to 2 representing flat to steep gradient; here slope was set at 1.2 throughout) (CSIRO 2007). LW gain (LWG) was calculated as ER/(0.92 · EVG), where ER refers to the MEI surplus to other needs (maintenance and grazing) and EVG refers to ME content of empty weight gain, which is based on several constants, total MEI and MEI for maintenance, and relative size (CSIRO 2007).

Enteric $CH_4$ emissions from the dairy heifers between weaning and first mating were estimated using the Australian National Greenhouse Gas Inventory (NGGI) methodology (DCCEE 2012). Since the NGGI method requires nutritive data additional to those used in the Feeding Standards for Australian Livestock, DM digestibility (DMD) was calculated on the basis of the equation:

$$\text{DMD (as a fraction)} = (\text{ME} + 1.037) / 0.1604 \text{ (DCCEE 2012).}$$

A global warming potential of 21 was used to convert $CH_4$ emissions into carbon dioxide equivalents ($CO_2e$; DCCEE 2012). Enteric $CH_4$ emissions of the heifer calves before weaning were not included in either the static or dynamic analyses, assuming that emissions before rumen maturation were negligible and not different between diet treatments (IPCC 2006).

### *6.2.2. Dynamic analysis*

A limitation of the static approach was that it assumed that heifers received constant feed supply throughout the year, resulting in consistent LWG over the period between weaning and mating. Clearly, this assumption is not realistic; so, we also undertook a mechanistic approach to estimate the time and GHG emissions between weaning and mating. In contrast to the static approach, the dynamic approach accounted for climate variation, both within and across years, changes in pasture quantity and nutritive value, and their compounding influence on animal growth rates and cumulative enteric $CH_4$ emissions.

The Sustainable Grazing Systems (SGS) whole-farm biophysical model (Johnson 2013, version 5.3), herein referred to as the SGS model, was used to simulate a 100ha dairy farm, with animals grazing rainfed kikuyu pastures all year and with some supplementary feeding to ensure that average annual dietary ME was similar to that used in the static analysis (see below). The SGS model is a whole-farm-system model that includes modules for soil water and nutrient flows, pasture production and utilisation, and animal intake and growth (Johnson 2013). The SGS model has been used extensively in Australia to model grazing enterprises for temperate and, to a lesser extent, subtropical farm systems (Lodge and Johnson 2007; Powell *et al.* 2011; Christie *et al.* 2014; Doran-Browne *et al.* 2014). Validation has shown that the model accurately estimates pasture consumption and seasonal DM accumulation (Cullen *et al.* 2008; Rawnsley *et al.* 2009; Doran-Browne *et al.* 2014).

Johnson *et al.* (2012) described the animal module where growth and energy dynamics is documented in response to available energy, and includes body protein, water, and fat components. The growth of protein is defined using a Gompertz equation, with protein weight taken to be the primary indicator of metabolic state. Fat is regarded as a potential source of metabolic energy for physiological process, such as the resynthesis of degraded protein. Fat growth is secondary and depends on current protein weight, as well as maximum potential fat fraction of bodyweight (BW), which varies through animal growth as defined by total BW. Protein is subject to turnover, so maintaining protein reserves requires the resynthesis of degraded proteins. This maintenance, along with energy required for activity, takes precedence over growth of new tissue. New growth of fat depends on current protein weight, as well as the maximum potential fat fraction of BW, with this maximum varying throughout the growth of the animal. Default animal BW characteristics, growth dynamics and energy dynamics are defined in Johnson *et al.* (2012).

In the SGS model, animal feed is assumed to comprise the following three basic components: neutral detergent fibre (NDF), which is primarily cellulose, hemicellulose and lignin in cell-wall material; protein; and the remainder, which is the neutral detergent solubles, is mainly sugars for pasture but may include compounds such as starch and fat for other feeds. It is assumed that, for all feed types, the digestibility of protein and neutral detergent solubles are fixed at 85%, so that total digestibility is a function of feed composition (pasture, supplement and forage) and the digestibility of the NDF component. The default digestibility of NDF for kikuyu is defined as being 60% under non-limiting water conditions and 30% for dead material (Johnson 2013). The digestibility of NDF varies linearly between these two values as the growth limiting factor for water varies between 1 and 0 (no water stress to full water stress) and influences the amount of live and dead material. The NDF of the overall canopy is, therefore, influenced by shoot and root growth and senescence, as well as the leaf and sheath components of shoot growth. The ME available to the animal for use in maintenance, growth and lactation is the difference between the gross energy of feed intake and energy costs associated with the production of $CH_4$, urine and dung.

The default kikuyu parameters in the SGS model (Bell *et al.* 2013b) were used in combination with daily weather data for Gympie (26.2°S, 152.7°E), which were accessed as a patched point dataset from the Bureau of Meteorology SILO dataset (Jeffrey *et al.* 2001). The climate-data inputs were between 1 January 1970 and 31 December 2013 and included daily minimum and maximum temperature (°C), rainfall (mm), solar radiation (MJ/m2), vapour pressure (kPa) and evapotranspiration (mm). Over the 44-year simulation period, the mean minimum and maximum temperatures were 14.1°C and 26.9°C, respectively. Rainfall was summer dominant and averaged 1099 mm/annum over the simulation period, varying between 600 and 1766 mm/annum. The soil type was classified as a uniform medium textured loam soil with a bulk density of 1.5 g/cm3, saturated hydraulic conductivity of 30 mm/h in the top 400 mm of soil profile, and saturation point, field capacity and wilting point of 43%, 29% and 17% by volume, respectively (Isbell 2002). Nitrogen fertiliser (N) was applied as urea annually during the growth phase of kikuyu, on 1 October, 1 December and 1 February, at a rate of 20 kg N/ha per application, representing kikuyu pastures with suboptimal management (Garcia *et al.* 2014).

Stocking rate and grazing rules for pre-grazing biomass and post-grazing residual are defined by the model user and, in the present study, a herd of 350 heifers entered the dairy farm as weaners with a LW of 100 kg. We aligned stocking rate with local expert knowledge for the region, so as to minimise the amount of excess pasture required to be conserved during the summer growth period. The model rules paddock selection based on available biomass and leaf stage and, in the study, the herd rotationally grazed 20 x 5 ha paddocks down to a residual biomass of 1.2 t DM/ha once the pastures reached the 4.5-leaf regrowth stage (Reeves *et al.* 1996), or before this, if the pasture biomass had reached 3.0 t DM/ha. These are aligned with best management practices for kikuyu pastures (Fulkerson *et al.* 2010). Animals were removed when the herd average reached 360 kg LW. The following 1 January, another herd of 350 weaned heifers entered the dairy-farm system, beginning the management cycle anew.

For simplicity, we chose to simulate two levels of average annual MEI (rather than the four levels as used in the static analysis), since the aim was to gauge the difference in time taken and associated GHG emissions in reaching mating LW when diet nutritive values varied over time. We conducted simulations by iteratively adjusting the quantity of supplement fed so that, over the 44-year simulation period, the mean annual diet ME was as close as possible to 9.5 and 11.0 MJ/kg DM as per the static approach. The lower level required minimal concentrate feeding and mainly consisted of kikuyu pasture, whereas the higher energy level required addition of both grain and forage to the pasture diet, so as to attain an average ME of 11 MJ/kg DMI. These two systems reflected field data for case-study farms 1 and 3, presented in the Supplementary Material in section 6.5.

The SGS model allows users to define the order of preference and the minimum and maximum intake of feeds across four broad categories (i.e. grazed pastures, concentrates, mixed ration and forages). This function was implemented in the study and details of feeding preference for the two diets are shown in Table 6.1. For the 9.5 MJ ME/kg DM diet, there was no pasture substitution because the pasture was allocated first, with substitution occurring only if the order of preference has concentrates before pasture. For the 11 MJ ME/kg DM diet, concentrate was first fed, then the partial mixed ration and finally all available pasture. Pasture substitution occurred only when the daily sum of concentrates, total mixed ration and pasture were above daily feed demand. The effect would have been additional pasture being available for conservation (cutting once the biomass reached 3.5 t DM/ha), as opposed to allowed to become mature, less digestible and restricting intake and/or leaving higher-than-optimum post-grazing residuals. Surplus conserved pasture was not used as a feed source in the current study.

Enteric $CH_4$ emissions were estimated on the basis of the fraction of digestible energy lost through $CH_4$ fermentation, which was 6.1% and 4.2% from high-fibre and low-fibre feed sources, respectively (Johnson 2013). For the current study, pastures represented a high-fibre feed source, whereas concentrates and partial mixed ration represented low-fibre feed sources. Enteric $CH_4$ emissions were then multiplied by 21 to convert into $CO_2$e (DCCEE 2012).

**Table 6.1** Feeding preference and diet composition for two heifer raising systems examined in the dynamic analysis using the SGS model (See text for details).

| **Feeding preference** | **Low quality pastured-based diet (9.5 MJ ME/kg DM)** | **Partial mixed ration diet (11.0 MJ ME/kg DM)** |
|---|---|---|
| 1st | Kikuyu pasture; quality and quantity estimated by SGS model | Concentrates; max. 1.5 kg DM/head.day: 12.0 MJ ME/kg DM |
| 2nd | Concentrates; max. 1.5 kg DM/head.day: 12.0 MJ ME/kg DM | Partial mixed ration; max 1.8 kg DM/head.day; 12.0 MJ ME/kg DM |
| 3rd | n.a. | Kikuyu pasture; quality and quantity estimated by SGS model, subject to climate and seasonal variation |

DM, dry matter; ME, metabolisable energy

### *6.2.3. Data analysis*

For each simulation, the number of days required to reach target LW was computed for each herd of heifers. Mean ME concentration of the diet (MJ ME/kg DM) for each herd was estimated by dividing the sum of daily MEI (MJ ME/head) by the sum of daily DM intake (kg DM/day) and averaging this across all the herds over the period that the heifers were present on the farm. For each simulation, the daily enteric $CH_4$ emission was summed to estimate cumulative herd $CH_4$ emissions between weaning and reaching the target mating LW. Overall, average days to target LW, MEI, ME concentration and enteric $CH_4$ emissions were then computed over the 44-year simulation period.

## **6.3 Results**

### *6.3.1. Static analysis*

The four diets strongly influenced the time required to reach the target mating LW. The highest energy diet of 11.0 MJ ME/kg DM resulted in a daily LWG of 1.1 kg per day, with the number of months between weaning and target mating LW being 8.0 (Table 6.2). This LWG equates to a heifer achieving target mating LW at ~11 months of age. When fed the lower-energy diet (9.5 MJ ME/kg DM), the estimated LWG was 0.6 kg per day and the number of months between weaning and target mating LW was 14.2 (Table 6.2). This LWG equates to a heifer achieving target mating LW at ~18 months of age. As diet quality improved, cumulative DMIs between weaning and target LW declined, such that cumulative enteric $CH_4$ was reduced from ~1.2 to 0.8 t $CO_2$e/heifer when diet quality improved from 9.5 to 11.0 MJ ME/kg DM, even though daily emissions per heifer did not vary with diet quality (Table 6.2). Alternatively, improving diet quality from 9.5 to 11.0 MJ ME/kg DM resulted in emission intensity being reduced from 4.6 to 3.0 kg $CO_2$e/kg LW (Table 6.2) as a result of the heifers reaching target LW for mating ~7 months sooner with the 11.0 MJ ME/kg DM diet than with the 9.5 MJ ME/kg DM diet (Table 6.2).

**Table 6.2** Estimated months to reach a target mating liveweight of 360 kg, average liveweight gain, total dry matter intake, energy intake and enteric emissions of a dairy heifer offered an *ad libitum* diet containing 100% $C_4$ grasses and with nutritive values of 9.5, 10.0, 10.5 or 11.0 MJ ME/kg DM for the static approach.

| Parameter | 9.5 | 10.0 | 10.5 | 11.0 |
|---|---|---|---|---|
| Months post-weaning to first mating | 14.2 | 11.4 | 9.4 | 7.9 |
| Average LW gain (kg/day) | 0.61 | 0.76 | 0.92 | 1.09 |
| Cumulative DMI (t DM/heifer) | 3.2 | 2.5 | 2.1 | 1.7 |
| Cumulative energy intake (GJ/heifer) | 30.0 | 25.3 | 21.8 | 19.2 |
| Cumulative enteric emissions (t $CO_2$e/heifer) | 1.2 | 1.0 | 0.9 | 0.8 |
| Emissions per day between weaning and mating (kg $CO_2$e/day) | 2.2 | 2.2 | 2.2 | 2.2 |
| Emissions intensity between weaning and mating (kg $CO_2$e/kg LW) | 4.6 | 3.9 | 3.4 | 3.0 |

$CO_2$e, carbon dioxide equivalent; DM, dry matter; GJ, Gigajoule; LW, liveweight

### *6.3.2. Dynamic analysis*

#### *6.3.2.1.* **Feeding animals a low-quality pasture-based diet (average 9.5 MJ ME/kg DMI)**

Over the 44-year simulation period, 18 herds of heifers were grown out to a LW of 360 kg on the low-quality pasture-based diet. Eleven herds took between 1 and 2 years post-weaning to reach the target LW. The remaining seven herds took between 2 and 3 years post-weaning as a result of periods of LW plateauing and/or loss when overall daily energy intake was compromised due to lower-than-optimum feed quality and/or availability. The time (mean ± s.d.) post-weaning required to reach the mating LW over the simulation period was 22.2 ± 4.0 months for an average diet ME of 9.5 MJ/kg DM (Table 6.3), with individual herds ranging between 16.5 and 27.7 months (Figure 6.1*a*). The mean MEI was 48.2 MJ ME/head.day on the basis of a mean daily intake of 5.1 kg DMI/heifer and a diet ME (mean ± s.d.) of 9.5 ± 0.1

MJ/kg DM. Mean cumulative enteric $CH_4$ emission over the simulation period was 1.2 t $CO_2$e/head (Table 6.3), with individual herds ranging between 1.0 and 1.5 t $CO_2$e/head (Figure 6.1*a*). Mean LWG was 0.4 kg/day, with each unit LWG resulting in an enteric $CH_4$ emission of 4.7 kg $CO_2$e (Table 6.3). For each additional day required to reach the target LW for first mating, there was an associated increase in enteric $CH_4$ emissions of 1.8 kg $CO_2$e/heifer (Table 6.3).

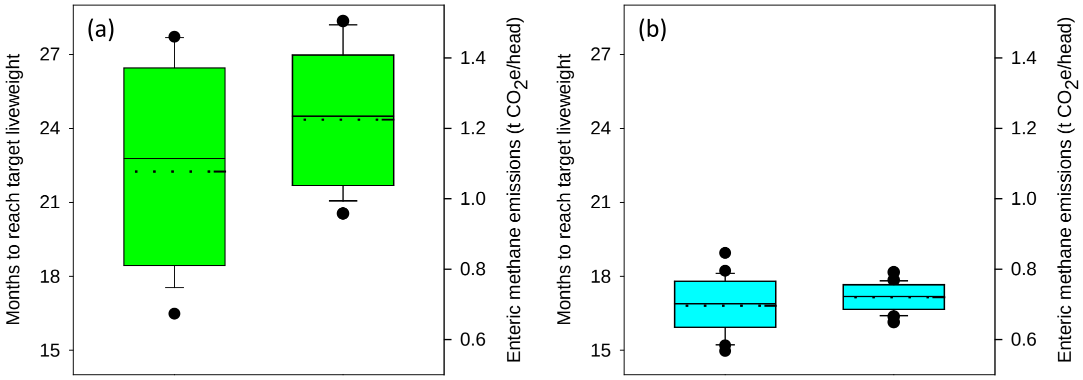


**Figure 6.1** Estimated number of months from weaning to target liveweight for mating (LHS axis) and associated enteric methane emissions (RHS axis) for heifers fed a diet with a mean metabolisable energy of (*a*) 9.5 MJ /kg dry matter intake ( ) and (*b*) 10.9 MJ/kg dry matter intake ( ), based on the dynamic approach. Boxes represent 25th and 75th percentiles, whiskers represent 10th and 90th percentiles, dots represent outliers, solid lines represent medians and dashed lines represent means.

**Table 6.3** Estimated months to reach a target mating liveweight of 360 kg, average liveweight gain, total dry matter intake (DMI), energy intake and enteric emissions of a dairy heifer offered a diet with either a mean metabolisable energy of 9.5 or 10.9 MJ/kg DMI for the dynamic approach.

| **Parameter** | **9.5 ± 0.1** | **10.9 ± 0.1** |
|---|---|---|
| Months post-weaning to first mating | 22.2 ± 4.0 | 16.8 ± 1.1 |
| Average LW gain (kg/day) | 0.40 ± 0.07 | 0.51 ± 0.03 |
| Cumulative DMI (t DM/heifer) | 3.4 ± 0.6 | 2.3 ± 0.1 |
| Cumulative energy intake (GJ/heifer) | 32.5 ± 5.4 | 25.6 ± 1.4 |
| Cumulative enteric emissions (t $CO_2$e/heifer) | 1.22 ± 0.20 | 0.72 ± 0.04 |
| Emissions per day between weaning and mating (kg $CO_2$e/day) | 1.82 ± 0.08 | 1.41 ± 0.02 |
| Emissions intensity between weaning and mating (kg $CO_2$e/kg LW) | 4.71 ± 0.75 | 2.77 ± 0.14 |

$CO_2$e, carbon dioxide equivalent; DM, dry matter; GJ, Gigajoule; LW, liveweight

Figure 6.2a illustrates the effect that seasonal climatic conditions had on heifers reaching the target LW, compared with the static approach when fed a diet with an average ME of 9.5 MJ/kg DM. The green line illustrates the static approach where heifers took 13.5 months post-weaning to reach the target LW. In comparison, the cyan and red lines illustrate two extremes of the dynamic approach, where it took 16.5 and 27.7 months post-weaning to reach the target LW (Figure 6.2a). As LW increased up to ~200 kg, there was little difference in enteric $CH_4$ emissions between the two dynamic examples. However, some herds experienced periods of very low total available energy intake due to low pasture quality and/or availability, resulting in LW loss that significantly extended the time required and cumulative $CH_4$ emissions before reaching the target LW (red line in Figure 6.2a).

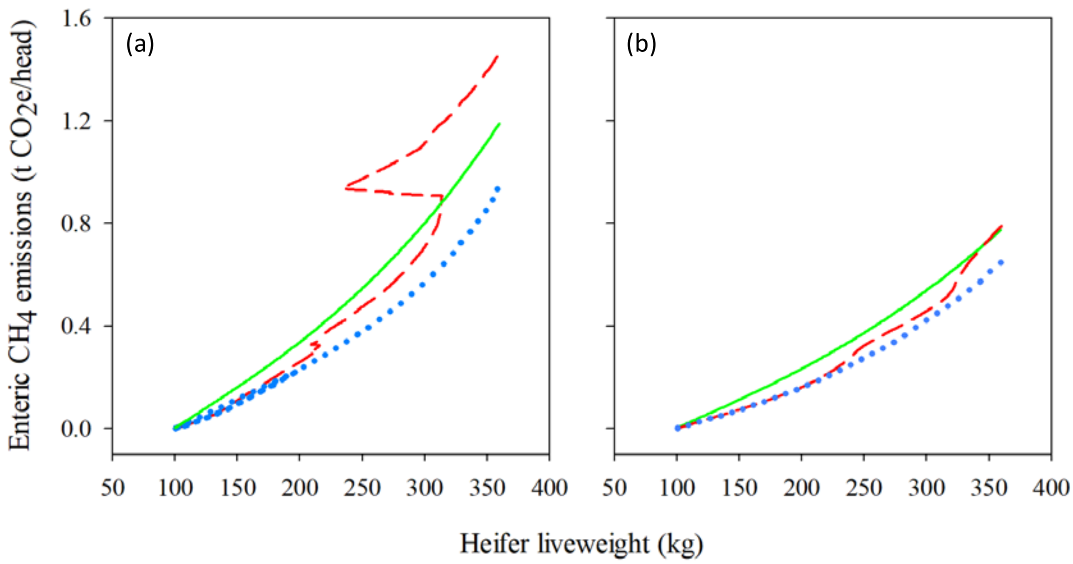


**Figure 6.2** Estimated cumulative enteric methane emissions (t carbon dioxide equivalents ($CO_2e$)/head) from weaning to mating when fed a diet with a mean metabolisable energy of approximately (*a*) 9.5 MJ/kg dry matter intake and (*b*) 10.9 MJ/kg dry matter intake. The solid green lines ( ) represent the static approach. The dotted cyan lines ( ) represent an example of a succinct period to reach the target liveweight with the dynamic approach, based on favourable climatic conditions. The dashed red lines ( ) represent an example of a prolonged period to reach the target liveweight with the dynamic approach, based on unfavourable climatic conditions.

### *6.3.2.2.* Feeding animals a high-quality partial-mixed ration diet (average 10.9 MJ ME/kg DMI)

Over the 44-year simulation period, all 22 herds of heifers were grown out to a LW of 360 kg on a high-quality ration, taking between 1 and 2 years post-weaning to reach the target LW. The time (mean ± s.d.) required to reach mating LW over the simulation period was 16.8 ± 1.1 months post-weaning (Table 6.3), with individual herds ranging between 15.0 and 18.9 months (Figure 6.1b). Over the simulation period, mean MEI was 50.1 MJ ME/head.day, mean daily intake was 4.6 kg DM and ME (mean ± s.d.) was 10.9 ± 0.1 MJ/kg DM. Mean enteric $CH_4$ emissions over the simulation period for the 11.0 ME diet were 0.7 t $CO_2e$/head (Table 6.3), with individual herds ranging between 0.6 and 0.8 t $CO_2e$/head (Figure 6.1b). Mean LWG was 0.5 kg/day, with each kilogram of LWG resulting in an enteric $CH_4$ emission of 2.8 kg $CO_2$e (Table 6.3). Heifers emitted 1.4 kg $CO_2$e every additional day required to reach the target LW (Table 6.3).

Figure 6.2b illustrates the effect that seasonal climatic conditions had on heifers reaching the target LW, compared with the static approach when the animals were fed a diet with an average ME of 10.9 MJ/kg DM. The green lines illustrates the static approach where heifers took 8.3 months post-weaning to reach the target LW. In comparison, the cyan and red lines illustrate two extremes of the dynamic approach, where it took 15.0 and 18.9 months post-weaning to reach the target LW, respectively (Figure 6.2b). Both dynamic-approach examples had lower cumulative enteric $CH_4$ emissions than did the static approach, indicating that while diet quality was similar to the static approach, average daily MEI may have been slightly greater. There was a divergence of LWG between the best and worst-case scenarios once the herds attained ~230 kg LW. This illustrates that there was little variation in feed quality and availability for these two herds before this stage, but, afterwards, variation in climatic conditions affected diet quality and/or availability of the pastures and, even with the inclusion of supplementary feed, there was still an influence on LWG.

## 6.4 Discussion

The present study explored the effects of dietary ME content and computation method on the time required for weaned heifers to reach the target LW for mating, and their resultant enteric $CH_4$ emissions, using the different methods. The first computation method was a static approach that assumed that feed supply was non-limiting and constant over time. LWG was a function of total energy intake due to feed quality and metabolic demand. Increasing the diet quality from 9.5 to 10.0 MJ ME/kg DM reduced the attainment of target LW at mating age by 84 days. Mating age was reduced by a further 60 and 44 days when diet quality increased from 10.0 to 10.5 and from 10.5 to 11.0 MJ ME/kg DM, respectively. For every 0.1 MJ ME/kg DM increase, enteric $CH_4$ emissions were reduced by ~55 kg $CO_2$e/heifer over the period between weaning and mating, resulting in a ‘dilution of maintenance’ of their lifetime GHG emissions (Bauman *et al.* 1985). Improving diet quality from 9.5 to 10.9 MJ/kg DM with the dynamic approach reduced enteric $CH_4$ emissions by 72 kg $CO_2$e/heifer over the period between weaning and mating. The dilution of emissions due to energy maintenance by increasing productivity is a key to carbon abatement on livestock farms (Steinfeld *et al.* 2006).

The primary reason for undertaking the dynamic analysis was to contrast the influence of climate on pasture availability and quality on LWG with a static approach. Total daily energy intake was a function of diet quality, pasture availability and animal metabolic state. At various times of the year, feed demand exceeded pasture supply, due to lower temperatures and lower soil moisture during the cooler months restricting plant growth (Bell *et al.* 2013b; Garcia *et al.* 2014). Over the 44-year duration of the study, mean annual DM production was 10.8 t DM/ha but annual DM varied between 7.4 and 16.4 t DM/ha, varying the time to reach the target LW for mating for the dynamic approach. Even with supplementation with concentrates and/or forages, the net result was a lower-than-optimum energy intake and, thus, little change or a decline in LW (red line in Figure 6.2a). This LW trend adequately mimics measured LW trends from field experiments in the region (Harrison *et al.* 2015). In reality, this would be a key period for management intervention with higher-quality alternative pastures and/or additional supplementary feeding to reduce the risk of LW loss. While farmers are likely to identify declines in LW, a stagnation of LWG is likely to be less obvious and highlights the importance of measuring and monitoring LWG on a regular basis so that LW targets at various ages are met (Dairy Australia 2003; Jagoe and Beggs 2013).

There was also a trend for lower feed quality during late spring through to mid-autumn, due to the case-study site being rainfed and a relatively low input of N fertiliser (Fulkerson *et al.* 1998, 2007). Improved feed quality and quantity, especially over the summer period, could be achieved with increased N fertilisation and/or irrigation (Kemp 1975; Marais 2001; Garcia *et al.* 2014). Over-sowing with other pasture species could also be implemented to increase production throughout the autumn to spring period when kikuyu growth declines (Bell *et al.* 2013b), thus maintaining heifers on higher-quality pastures year round to minimise the loss of LW over this period. In a more temperate environment and under irrigation, Botha *et al.* (2008) found that over-sowing an existing sward of kikuyu with either white (*Trifolium repens*) and red (*Trifolium pratense*) clover or with annual ryegrass (*Lolium multiflorum* var. *westerwoldicum*) was able to increase annual DM production by 5% and 14%, respectively, with most of this increased production occurring in spring. A beneficial outcome of the companion species was to increase summer ME values by 2.5 and 1.3 MJ/kg DM for the kikuyu–clover and kikuyu–annual ryegrass swards, respectively, relative to the kikuyu monoculture sward. This increase in ME continued into autumn for both over-sown kikuyu swards and should be researched further under Australian subtropical conditions.

The estimated age to first mating on the low-quality diet was 22.2 ± 4.0 months with the dynamic approach. This period was considerably longer than the 14.2 months estimated with the static modelling approach for a diet with the same energy content but assumed *ad libitum* daily intake. When climatic conditions were more optimal for kikuyu production, the period to mating could be reduced to as low as 16.8 months, a result that was slightly higher than that from the static approach but similar to values seen in southern states of Australia for heifers grazing temperate pastures. This highlighted the benefits of dynamic modelling approaches that account for the inter- and intra-annual variation in seasonal pasture supply.

Similarly, the cumulative estimated enteric $CH_4$ emissions were similar between the static and dynamic approaches when the diet was increased to an ME of ~11 MJ/kg DM at ~0.8 t $CO_2$e/heifer (Figure 6.2b). However, the time to reach the target LW was more than double with the dynamic approach compared with the static approach. In addition, when comparing the timeframes when climatic conditions were more optimum, the period to mating was at best reduced to ~15 months, which was substantially greater than the 7.9 months estimated with the static approach. This separation between the two approaches as feed quality improved highlighted the importance of dynamic modelling to estimate the range of potential outcomes in times to reach the target LW due to climatic and nutritional variation within and among years.

While the focus of the study was the effect of diet quality on LWG and enteric $CH_4$ emissions, we also explored the effect of N intake on $N_2O$ emissions. In a well-managed kikuyu pasture, protein is generally considered non-limiting to LWG (Reeves *et al.* 1996; Marais 2001). For the static approach, N concentration of the diet was assumed to be non-limiting and retained as a constant over the period from weaning to mating, obviating the need to examine $N_2O$ emissions for the static approach. For the dynamic approach, the influence of climatic conditions on diet quality and availability resulted in varying N intakes and outputs, in turn resulting in variation in $N_2O$ emissions between the two nutritive-value diets. However, averaged over the period from weaning to mating, mean $N_2O$ emissions were 0.09 and 0.08 t $CO_2$e/head (based on $N_2O$ emissions 310 times the value of $CO_2$ (DCCEE 2012)) for the 9.5 and 10.9 ME diets, respectively, representing 7% and 11% of enteric $CH_4$ emissions, respectively (see Table 6.2). The results are consistent with those of the past studies of livestock emissions when grazing subtropical pastures (Harrison *et al.* 2015, 2016) where small $N_2O$ values relative to $CH_4$ justify the focus of the present study on enteric $CH_4$ emissions only.

One aspect of the environment not included in the dynamic modelling was the effect of stress on LWG. Any set-back in growth due to stress, such as adverse seasonal conditions (e.g. high summer heat), parasites or disease will delay reaching daily LWG targets (Moss 1993; Le Cozler *et al.* 2008). For tropical farm systems, such as those explored in the current study, heat stress can have a substantial effect on feed intake (Marcillac-Embertson *et al.* 2009; Gaughan *et al.* 2014). The present study did not simulate heat-stress effect on DMI and subsequent LWG, and this may have further delayed heifers reaching the target LW for mating, beyond what was estimated here.

### 6.5 Conclusion

The current study has illustrated that any impediment to heifer LWG has GHG emission implications. When optimal LWG is not achieved, mating is delayed and the animal spends a greater proportion of its life not producing milk while still emitting GHGs. One reason for these delays can be attributed to lower-than-optimum feed quality and energy intake. The static modelling approach in the current study showed that for heifers in subtropical environments, a diet with a minimum ME of 9.5 MJ ME/kg DM is required for heifers to reach the target LW for mating, so as to calving at 2 years of age. However, this assumes that the heifers are fed to *ad libitum* intake on a daily basis to maximise MEI. This has the potential to not be the case in reality. The present study showed that inputs of high-quality supplementary feed during periods of low quality and/or quantity of subtropical pastures are required to reduce mating age for heifers to calve nearer to the optimum 24 months of age, to maximise lifetime milk production. This study also showed that although estimation of enteric $CH_4$ emissions was similar between the static and dynamic modelling, rates of LWG differed substantially between the two approaches, resulting in significant differences in the time required to reach the mating LW. With increasing pressure for the Australian dairy industry to reduce its carbon footprint, the present study emphasises the importance of increasing growth rates of heifers, with the most important being feeding animals with energy-dense feedstuffs. Together, these factors reduce overall feed requirements and cumulative emissions of enteric $CH_4$ before mating, to reduce the emissions intensity of milk production over the lifetime of the animal.

### 6.6 Acknowledgements

The authors acknowledge insights provided by W.J. Fulkerson. We are grateful for the support of the three farmers who supplied case-study data to formulate the modelled farm systems. We are also grateful for support from their host organisations and funding from the Australian Department of Agriculture, Dairy Australia, Meat & Livestock Australia and Australian Wool Innovation.

### 6.7 Supplementary material

Age at first mating and calving data were obtained from three farms within the subtropical dairying region of northern New South Wales and southern Queensland, Australia. This region was selected because it has traditionally had the highest age at first calving across the eastern seaboard of Australia (Hough 1992). We selected three farms as representative of one of three alternative heifer-raising systems.

- Case study farm 1- Dryland tropical pastures and minimal concentrates (Gympie, QLD; 26.2°S, 152.7°E)

The ME concentration of the diet was estimated at 9 MJ/kg DM over the duration between weaning and mating (Trevaskis, unpub. data. 2014). Between weaning at 3 months of age and 12 months of age, heifers grazed rainfed $C_4$ pastures (predominantly Setaria (*Setaria anceps*) with some Bisset Bluegrass (*Bothriochloa insculpta*) and Kikuyu (*Pennisetum clandestinum*); *c*. 8-9 MJ ME/kg DM). Heifers were fed concentrates (*c*. 12.0 MJ ME/kg DM) at a rate of 1 kg DM/head.day. Heifers also grazed annual ryegrass (*Lolium rigidum*) (*c* 11.5 MJ ME/kg DM) over winter when the dairy farm had surplus annual ryegrass pastures to milker requirements. Between 12 months of age and mating at 23 months of age, heifers grazed the same pasture species as mentioned above with concentrate feeding of 1.7 kg DM/head.day.

- Case study farm 2- Supplements and grazing (Casino, NSW; 28.8°S, 153.0°E)

The ME concentration of the diet was estimated at 11 MJ/kg DM over the duration between weaning and mating (Trevaskis, unpub. data. 2014). Between weaning at 3 months of age and 9 months of age, the heifers were fed perennial ryegrass (*Lolium perenne*) silage (9.5 MJ ME/kg DM) and ad libitum concentrates containing 90% triticale and 10% canola meal (*c* 13.2 MJ ME/kg DM). Between 9 and 14 months of age, heifers grazed annual ryegrass (*c* 10- 12 MJ ME/kg DM) or setaria (*c* 10 MJ ME/kg DM) and received 2.2 kg DM concentrate/head.day. Between 14 months of age and mating at 17.5 months of age, heifers grazed either annual ryegrass or setaria and received no concentrates.

- Case study farm 3- Grazed pastures and supplementary feeding (Gympie, QLD; 26.2°S, 152.7°E)

The ME concentration of the diet was estimated at 11 MJ/kg DM over the duration between weaning and mating (Trevaskis, unpub. data. 2014). Between weaning at 3 months of age and 6 months of age, heifers grazed rain-fed kikuyu (*c* 10 MJ ME/kg DM) and were fed ad libitum hay (*c* 8 MJ ME/kg DM) and calf-pellet mix at a rate of 2 kg DM/head.day (*c* 12.5 MJ ME). Between 6 months of age and mating at 17 months of age, heifers grazed rain-fed kikuyu and had ad libitum access to a heifer mix which was predominantly palm kernel extract (*c* 11 MJ ME/kg DM).

# CHAPTER 7 REVISED GREENHOUSE-GAS EMISSIONS FROM AUSTRALIAN DAIRY FARMS FOLLOWING APPLICATION OF UPDATED METHODOLOGY

**CSIRO** PUBLISHING

*Animal Production Science*, 2018, **58**, 937–942
http://dx.doi.org/10.1071/AN16286

## Revised greenhouse-gas emissions from Australian dairy farms following application of updated methodology

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**Abstract.** Every year since 1990, the Australian Federal Government has estimated national greenhouse-gas (GHG) emissions to meet Australia's reporting commitments under the United National Framework Convention on Climate Change (UNFCCC). The National Greenhouse Gas Inventory (NGGI) methodology used to estimate Australia's GHG emissions has altered over time, as new research data have been used to improve the inventory emission factors and algorithms, with the latest change occurring in 2015 for the 2013 reporting year. As measuring the GHG emissions on farm is expensive and time-consuming, the dairy industry is reliant on estimating emissions using tools such as the Australian Dairy Carbon Calculator (ADCC). The present study compared the emission profiles of 41 Australian dairy farms with ADCC using the old (pre-2015) and new (post-2015) NGGI methodologies to examine the impact of the changes on the emission intensity across a range of dairy-farm systems. The estimated mean (±s.d.) GHG emission intensity increased by 3.0%, to 1.07 (±0.02) kg of carbon dioxide equivalents per kilogram of fat-and-protein-corrected milk (kg $CO_2$e/kg FPCM). When comparing the emission intensity between the old and new NGGI methodologies at a regional level, the change in emission intensity varied between a 4.6% decrease and 10.4% increase, depending on the region. When comparing the source of emissions between old and new NGGI methodologies across the whole dataset, methane emissions from enteric fermentation and waste management both increased, while nitrous oxide emissions from waste management and nitrogen fertiliser management, $CO_2$ emissions from energy consumption and pre-farm gate (supplementary feed and fertilisers) emissions all declined. Enteric methane remains a high source of emissions and so will remain a focus for mitigation research. However, these changes to the NGGI methodology have highlighted a new 'hotspot' in methane from manure management. Researchers and farm managers will have greater need to identify and implement practices on-farm to reduce methane losses to the environment.

**Additional keywords:** carbon dioxide, ADCC, methane, nitrous oxide.

Received 4 May 2016, accepted 26 September 2016, published online 16 November 2016

## 7.1 Introduction

A rising global population, coupled with a strong demand for animal-sourced protein from an emerging global middle class, will increase demand for meat and milk products. By 2050, the demand for meat and milk is projected to more than double from 2010 levels (FAO 2011; Gerber *et al.* 2013a). There is an increasing recognition of the need to improve agricultural productivity and efficiency to meet this growing demand, while minimising environmental impacts. The Australian dairy industry is one of Australia's major rural industries, ranked third behind beef and wheat, producing 9.7 billion litres of milk from 1.7 million cows on 6100 farms (Dairy Australia 2016). In 2013–2014, the Australian dairy industry contributed ~9 Mt of carbon dioxide equivalents ($CO_2$e)/annum, equivalent to 11% of the nation's agricultural emissions (DoE 2015a).

The National Greenhouse Gas Inventory (NGGI) methodology used to estimate Australia's greenhouse-gas (GHG) emissions has altered over time, with the latest change occurring in 2015 for the 2013 reporting year. These changes were required to:

(1) ensure that the Australian inventory continues to meet international reporting requirements under the United Nations Convention on Climate Change and Kyoto Protocol,

(2) reflect the results of recent Australia-specific research, and

(3) signify changes to animal and feed characteristics and waste and crop-management practices that have occurred over time.

As measuring GHG emissions on farm is expensive and time-consuming, the dairy industry is reliant on estimating these emissions using tools such as the Australian Dairy Carbon Calculator (ADCC; formerly referred to as the Dairy Greenhouse Gas Abatement Strategies (DGAS) calculator). In 2012, 41 Australian dairy farms from diverse geographic locations, varying herd and farm sizes and levels of milk production per cow were examined for their GHG emissions (Christie *et al.* 2012). The current study has used this same previously published dataset to ascertain the effect of changes in NGGI algorithms and emission factors (EF) on total farm GHG emissions and emissions intensity (EI; annual total farm GHG emissions divided by annual farm milk production), regional EI and proportion of GHG emissions from each source. In addition, the current study examined the agreement between old and new NGGI methodologies with respect to individual-farm and regional EIs and the proportion of total GHG emissions from each source by applying a concordance correlation coefficient assessment.

## 7.2 Materials and methods

### *7.2.1. Dairy farm systems*

The present study used farm data from a previous study by Christie *et al.* (2012; Chapter 4), in conjunction with the updated 2015 Australian NGGI methodology (DoE 2015a), to estimate the updated whole-farm system EI of milk production. Forty-one dairy farms, as part of the Accounting4Nutrients (A4N) project, were selected using a stratified-random process, taking into consideration key criteria of (1) geographical location, (2) litres of milk per grazed hectare, (3) grazed hectares and (4) proportion of grazed hectares that were irrigated (Gourley *et al.* 2012b). Farms were selected from all states of Australia and were representative of their local industry. A summary of farm, herd and milk-production data can be found in Tables 7.1, 7.2 and 7.3, respectively. Herd sizes varied between 62 and 1350 milkers, with an overall average herd size of 355 milkers per farm. Total annual farm milk production varied between 0.4 and 11.2 million litres, with a mean of 2.2 million litres per annum. Milk production per cow varied between 2680 and 9150 L/cow.lactation (overall mean of 6030 L/cow.lactation). Milking platform size varied between 52 and 460 ha, with a mean of 192 ha. Nitrogen (N) fertiliser inputs varied between 0 and 316 kg N/ha (overall mean of 75 kg N/ha), and the proportion of the total farm irrigated, including outblock and runoff areas, varied between 0% and 82% (overall mean of 23%).

**Table 7.1** The mean (minimum and maximum in parenthesis) farm values required to estimate greenhouse gas emissions.

| Key farm input data | Mean (min and max) |
|---|---|
| Farm area- total (ha) | 338.6 (67.3 – 1045.6) |
| Farm area- milking platform (ha) | 191.7 (52 – 460) |
| Farm area- irrigated (ha) | 63.2 (0 – 329) |
| Farm area- non-irrigated (ha) | 128.6 (3 – 460) |
| Electricity (000’s kWh/yr) | 145.8 (27.2 – 1023.1) |
| Diesel (000’s L/annum) | 9.6 (6.2 – 25.4) |
| N fertiliser (000’s kg N/yr) | 23.4 (0.0 – 154.3) |
| P fertiliser (000’s kg P/yr) | 4.4 (0.0 – 25.1) |
| K fertiliser (000’s kg K/yr) | 8.5 (0.0 – 64.4) |
| S fertiliser (000’s kg S/yr) | 4.1 (0.0 – 26.0) |
| Purchased concentrates (t DM/yr) | 436.1 (19.9 – 2336.6) |
| Purchased forage (t DM/yr) | 233.4 (0.0 – 1788.7) |
| Purchased other feeds (t DM/yr) | 132.7 (0.0 – 2375.9) |

$CO_2e$, carbon dioxide equivalent; DM, dry matter

**Table 7.2** The mean (minimum and maximum in parenthesis) herd values required to estimate greenhouse gas emissions.

| **Key herd input data** | **Mean (min and max)** |
|---|---|
| Milking herd size (number of cows)[a] | 355 (62 – 1350) |
| Milking herd average liveweight (kg) | 534 (453 – 550) |
| Heifer herd size (number of rising 1 and 2 yr olds) | 72 (14 – 190) |
| Replacement rate (%) | 22.1 (3.9 – 36.1) |
| Mature bulls herd size | 7 (0 – 40) |
| Number of bulls per 100 milkers | 2.0 (0.0 – 6.6) |
| Stocking rate (cows/ha) | 2.0 90.6 – 4.4) |
| Pasture consumption (t DM/ha) [b] | 6.5 (0.1 – 14.1) |
| Concentrates (t DM/cow.lactation) | 1.3 (0.0 – 2.9) |
| Estimated total DMI (t DM/cow $lactation^{-1}$) [b] | 5.8 (3.6 – 7.8) |
| Dietary dry matter digestibility (%) | 74.5 (68.9 – 78.9) |
| Dietary crude protein (%) | 19.8 (14.4 – 24.4) |
| Feed conversion efficiency (litres of milk/kg DMI) | 1.04 (0.55 – 1.56) |
| Percentage of grain in the milking herd diet | 22.3 ( 0.0 – 57.4) |

[a] Cows milked for more than 2 months and contributing to annual milk production.

[b] Total dry matter intake from home-grown pasture, conserved pasture, purchased forage, purchased grain/concentrates and purchased other feed sources (t DM intake/cow/lactation), as calculated using the Pasture Consumption and Feed Conversion Efficiency Calculator (Heard and Wales 2009).

DM, dry matter; DMI, dry matter intake

**Table 7.3** The mean (minimum and maximum in parenthesis) milk production values required to estimate greenhouse gas emissions.

| Key milk production input data | Mean (min and max) |
|---|---|
| Milk production (000's litres/year) | 2183 (373 – 11248) |
| Milk production (000's kg MS/year) | 160 (26 – 780) |
| Milk production (000's kg FPCM/year) | 2254 (374 – 11067) |
| Milk production (000's kg FPCM/cow.lactation) | 6.27 (3.25 – 9.87) |
| Milk production (000's kg FPCM/ha milking platform) | 12.67 (3.20 – 36.05) |
| Milk production (000's kg FPCM/ha total farm) | 7.62 (1.41 – 18.08) |
| Annual mean butterfat (g/100 g milk) | 4.10 (3.65 – 5.13) |
| Annual mean protein (g/100 g milk) | 3.31 (3.09 – 3.83) |

MS, milksolids; FPCM, fat and protein corrected milk

Farms were visited quarterly over a 12-month period between February 2008 and February 2009 to gather data and feed samples (Gourley *et al.* 2012b). Data included most aspects required for assessing the GHG emissions, with an indirect estimation of electricity and diesel consumption by the authors (Christie *et al.* 2012). A representative sample of each feed source (pasture and supplementary feed) fed to the milking herd on the day of the visit was collected, prepared and analysed for dry-matter digestibility and crude protein, the two key input data for estimating methane ($CH_4$) and nitrous oxide ($N_2O$) emissions. As all of the farms reviewed in the present study were long-term established farms, there was no incorporation of carbon removal through land-use change such as deforestation. In addition, there was no incorporation of changes in soil carbon as this was beyond the scope of the original A4N project that captured farm data to allow for GHG-emission estimations.

### *7.2.2. Changes to the methodology*

There were several changes made to the methodology, which were incorporated into ADCC (version 4.2). The most significant was the alteration of the global warming potential (GWP) of $CH_4$ and $N_2O$. The GWP of $CH_4$ was increased from 21 to 25, while the GWP of $N_2O$ decreased from 310 to 298. This aligns the national GWPs with the revised UNFCCC reporting guidelines and the 2006 Intergovernmental Panel on Climate Change guidelines (IPCC 2006). Other major changes have been the simplification of enteric $CH_4$ emission estimation (kg $CH_4$/head.day) to 20.7 x intake (kg DM/head.day)/1000, on the basis of Australian measurement data (Charmley *et al.* 2016).

The allocation of waste (faeces and urine) to different manure management systems (MMS; namely pastures, anaerobic lagoon, daily spread from a sump system, draining daily to paddocks and solid storage) was altered using expert judgement based on Dairy Australia's Natural Resource Management surveys between the Years 2000 and 2012 (Dairy Australia 2012), with the proportion of waste allocated to pasture during grazing declining in all regions of Australia (Reyenga *et al.* 2015). Along with the change in the allocation of waste to the different MMS, the $CH_4$ conversion factor (MCF) has been altered for each MMS on the basis of regional mean temperatures. This has resulted in changes in the integrated MCF, calculated by multiplying the proportion of manure allocated to each MMS by its corresponding MCF, and summing these together (i.e.(% manure to lagoon x it's corresponding state MCF) + (% manure to daily spread from a sump system x it's corresponding state MCF) + (% manure to daily drained to paddock x it's corresponding state MCF) + (% of manure to solid storage x it's corresponding state MCF) + (% manure onto pasture x it's corresponding state MCF)). Most states in Australia use the same MCF for the majority of the MMSs, with the exception of anaerobic lagoon where the variation in regional daily temperature across the country resulted in MCFs varying between 0.70 and 0.77. However, given that the proportion of waste allocated to each MMS varies among states, the resultant is regional variation in integrated MCFs. For example, the integrated MCF with the old methodology was 0.065 for the three largest milk-producing states of Victoria, New South Wales and Tasmania. With changes to the integrated MCF factors as discussed here, the integrated MCF is now 0.958, 0.1016 and 0.067 for these three abovementioned states, respectively (DoE 2015a).

Several aspects of $N_2O$ estimations have also been altered. These have included the alteration of the N-to-protein conversion factor from 6.25 to 6.38, to better align with the protein concentration of milk as opposed to forages (IDF 2006). Changes to various EFs have also been undertaken to align with changes to IPCC default EFs that have occurred since the original NGGI methodology was developed in 1990 (DoE 2015a). These have included the following:

(1) EF for ammonia volatilisation changed from 1.0% to 0.4%,

(2) EF for direct N loss from faeces reduced from 0.5% to 0.4%, and

(3) reduction in the amount of N lost through leaching and runoff from 1.25% to 0.75%.

### *7.2.3. Statistical analyses*

Statistical program for the social sciences statistics (SPSS, IBM Corporation 2013) was used to perform a one-way ANOVA procedure, to determine the influence of region on EI. The Concordance Correlation Coefficient (CCC; Lin 1989, 2000),

using syntax by Marta Garcia-Granero (2005; http://gjyp.nl/marta/Lin.sps ; verified 25 Jan 2019), was used within SPSS to assess the agreement between old and new methodologies with respect to individual-farm EI, regional EI and proportion of total GHG emissions from each source.

## 7.3 Results

The updated NGGI methodology resulted in the mean estimated EI of milk production increasing from 1.04 to 1.07 kg $CO_2$e/kg fat- and protein-corrected milk (FPCM) (Table 7.4) across the dataset. The extent of agreement between methodologies, as estimated with the CCC, was 0.924 when comparing individual-farm EI. Individual-farm EI varied between 0.84 and 1.54 kg $CO_2$e/kg FPCM. When comparing the EI between the old and new NGGI methodologies at a regional level, EI increased for most regions. However, there was an overall mean reduction in EI for two of the eight regions, with Tasmania reducing by 4.6% and South Australia by 0.2% (Table 7.4). The extent of agreement between methodologies was 0.845 when comparing regional EI. The positive linear relationship between milk production and GHG emissions increased from 0.89 to 0.91 kg $CO_2$e/kg FPCM ($R^2$ = 0.96) with the change in methodology.

**Table 7.4** Estimated regional mean greenhouse-gas emission intensity (kg $CO_2$e/kg fat- and protein-corrected milk, FPCM) using the original methodology (up until 2014) and updated methodology (from 2015 onwards) and the percentage change between methodologies.

| Region | Emissions intensity (kg $CO_2$e/kg FPCM) | | % change |
|---|---|---|---|
| | Original methodology | Updated methodology | |
| New South Wales | 1.06[b] | 1.07[ab] | 1.5 |
| Queensland | 1.11[ab] | 1.13[ab] | 2.1 |
| South Australia | 0.99[b] | 0.99[b] | -0.2 |
| Tasmania | 1.30[a] | 1.24[a] | -4.6 |
| Northern Victoria | 0.94[b] | 0.98[b] | 4.6 |
| South-eastern Victoria | 1.00[b] | 1.03[b] | 3.6 |
| South-western Victoria | 0.93[b] | 1.02[b] | 9.6 |
| Western Australia | 1.02[b] | 1.12[ab] | 10.4 |
| Mean | 1.04 | 1.07 | 3.0 |

Within column values followed by different letters are significantly different at $P$ = 0.05

$CO_2$e, carbon dioxide equivalent; FPCM, fat and protein-corrected milk

The contribution of the various GHG emission sources, as a percentage of total farm GHG emissions, increased for enteric and waste $CH_4$ by 2% and 109%, respectively (Table 7.5). Nitrous oxide emissions, as a proportion of total farm GHG emissions, reduced dramatically, especially for indirect $N_2O$ losses due to a reduction in the EFs for the proportion of N lost through leaching and/or runoff and atmospheric volatilisation being converted into $N_2O$. The total GHG emissions from some sources, such as from purchased supplementary feeds did not change, but with the overall increase in total farm GHG emissions, the contribution of these to total farm GHG emissions declined (Table 7.5). Carbon dioxide from fertiliser production increased by 7% due to a change in the estimation of $CO_2$e emissions from the production of non-N-based fertilisers within ADCC, as opposed to any NGGI alterations (Table 7.5). The extent of agreement between the two methodologies was 0.990 when comparing the proportion of total farm GHG emissions from each source.

**Table 7.5** Estimated proportion of total farm greenhouse-gas (GHG) emissions from each source averaged over all 41 farms using the original methodology (up until 2014) and updated methodology (from 2015 onwards) and percentage change between methodologies.

| Source | Proportion of total farm GHG emissions | | % change from original methodology and GWP |
|---|---|---|---|
| | Original methodology and GWP | Updated methodology and GWP | |
| Enteric $CH_4$ (%) | 55.5 | 56.8 | +2 |
| Waste $CH_4$ (%) | 4.7 | 9.9 | +109 |
| Direct $N_2O$ from animal waste (%) | 6.6 | 6.3 | -4 |
| Indirect $N_2O$ from animal waste (%) | 8.4 | 4.4 | -48 |
| Direct $N_2O$ from N fertilisers (%) | 1.8 | 1.7 | -5 |
| Indirect $N_2O$ from N fertilisers (%) | 2.1 | 1.1 | -47 |
| $CO_2$ from fuel and electricity (%) | 9.6 | 8.9 | -8 |
| $CO_2$ from purchased grains/ concentrates (%) | 6.0 | 5.7 | -6 |
| $CO_2$ from purchased forages (%) | 2.4 | 2.2 | -9 |
| $CO_2$ from purchased fertilisers (%) | 2.9 | 3.1 | 7 |

$CH_4$, methane, $CO_2$; carbon dioxide, GWP; global warming potential; $N_2O$, nitrous oxide

## 7.4 Discussion

The increase in EI of milk production by ~3% across the 41 farms previously assessed for their GHG emissions using the older NGGI methodology (Christie *et al.* 2012) was driven by an increase in the GWP of $CH_4$ from 21 to 25 and changes in the allocation of animal waste to the various MMSs. Although the GWP of $N_2O$ emissions declined from 310 to 298, this decline did not compensate for the increase in $CH_4$ emissions, especially given that total $CH_4$ emissions were two-thirds of total farm GHG emissions. This was more evident by just considering on-farm GHG emissions (sum of $CH_4$ and $N_2O$ emissions) and excluding emissions that occur off-farm such as $CO_2$ emissions from energy consumption and pre-farm gate embedded emissions of purchased feeds. The change in $CH_4$ estimations within the updated NGGI methodology resulted in on-farm GHG emissions increasing by 10.6%. In contrast, the changes to $N_2O$ estimation reduced on-farm GHG emissions by 6.3%. The net result was an increase in on-farm GHG emissions of 4.2%.

The allocation of animal waste to the various MMSs resulted in a doubling of the proportion of total farm GHG emissions attributed to waste $CH_4$ emissions. The proportion of waste deposited onto pastures while grazing declined from the historical mean of 92% for most regions, to between 79% and 82% for all regions, with the exception of Tasmania declining to 85% (DoE 2015a). Therefore, a greater proportion of waste is now allocated to the various MMSs, such as anaerobic lagoons or daily spread via draining to paddocks. The result of this is an increase in the integrated MCF from 6.5% to 9.6% and 10.2% for two largest dairying regions of Victoria and New South Wales, respectively. The integrated MCF for Western Australia increased three-fold from 2.8% to 8.9% with the changes in NGGI methodology, resulting in the mean EI of milk production for this region increasing more than for any other region (Table 7.4). The EI of milk production reduced slightly in Tasmania, as the change in the proportion of waste to the various MMSs altered only slightly (0.2%), increasing the integrated MCF by to 6.7% (DoE 2015a).

A CCC analysis was used to estimate the extent of agreement between the old and new methodologies with respect to individual-farm EI, regional EI and proportion of total GHG emissions from each source. While there is little literature giving a descriptive scale for the degree of agreement for this analysis, values of >0.9 are considered to have moderate agreement, and values >0.99 have almost perfect agreement (McBride 2005). Using this scale, there was moderate agreement between methodologies for estimating individual-farm EI and almost perfect agreement between methodologies for estimating the proportion of GHG emissions from each source. There was poor agreement between methodologies for estimating regional EI. This was predicted to occur because regional differences in the proportion of manure to various management systems resulted in varying $N_2O$ emissions beyond the changes to equations and GWP that occurred for all farms. For example, in the previous NGGI methodology, the three largest producing states of Victoria, New South Wales and Tasmania allocated 92% of total waste deposited onto pastures during grazing. However, with the new NGGI methodology, the proportion of waste deposited onto pastures during grazing was reduced to 81.9%, 79.2% and 85.2% of total waste for Victoria, New South Wales and Tasmania, respectively. This divergence of the proportion of waste deposited onto pastures during grazing, and subsequent $N_2O$ emissions, caused some variation in emissions from these states and, thus, has been shown with the lower CCC for the regional average EI of milk production.

### 7.5 Conclusion

In 2012, the Australian dairy industry launched a whole-of-industry Sustainability Framework with 11 targets, including reducing GHG EI, across on-farm and manufacturing, by 30% by the year 2020 (Dairy Australia 2015b). The manufacturing sector is aiming to achieve its reduction target through reduced energy consumption. The farming sector is implementing a range of concurrent and complementary mitigation options, including improving cow productivity, feed quality, herd fertility and nutrient management to continue reducing its carbon footprint. However, difficulty in developing a baseline for the on-farm sector has limited quantification of these mitigation strategies across the industry.

This analysis has highlighted that a new ‘hotspot’ has emerged. With herd sizes having increased over time, the time cattle spend grazing pastures has generally declined, resulting in more waste being deposited onto ‘hard’ surfaces and, subsequently, handled in systems where the risks of losses of GHGs to the environment are greatest. Therefore, waste $CH_4$ emissions have emerged as an area that will need consideration in terms of mitigation options moving forward. In addition, the revised 100-year GWP for $CH_4$ has increased to 28 (IPCC 2013), which would also result in waste $CH_4$ being a larger contributor to an overall emission profile of a dairy system.

# CHAPTER 8 GENERAL DISCUSSION

## 8.1 Global issues and policy

The world is currently facing many challenges, including how we can simultaneously feed an increasing global population while reducing GHG emissions. The current population of 7.6 billion people is predicted to increase to 9.8 billion people by 2050 and possibly up to 13.2 billion people by 2100 (UNDESA-PD, 2017). More than half of the anticipated growth is expected to occur in Africa, with Asia expected to be the second largest contributor to this growth (UNDESA-PD, 2017). In addition to a global population increase, there has been a per capita increase in meat and milk consumption which is predicted to continue increasing, especially from the burgeoning rise in the ‘middle-class income’ population in the developing economies of the world (Delgado, 2003; Steinfeld *et al.*, 2006; Alexandratos and Bruinsma, 2012). By 2050, FAO (2011) estimates that there will be a 73% increase in meat and egg consumption and a 58% increase in dairy consumption worldwide compared with 2010 consumption data, with most of this increase occurring in the developing world (Delgado *et al.*, 1999).

At the UNFCCC’s COP21 Agreement meeting, 195 countries adopted the first universal, legally binding global climate plan. The agreement included the plan of avoiding dangerous climate change by limiting global warming to below 2°C, and if possible, below 1.5°C (European Commission, 2015). The Paris Agreement also stipulated anthropogenic emissions by sources will need to be balanced with sinks by the second half of the 21st Century to remain under this threshold (UNFCCC, 2015). Many of these countries made commitments to reducing GHG emissions from their agricultural sector, with 61 specifically mentioning livestock emissions as a site of action (Richards *et al.*, 2016).

Many countries of the world have set targets towards a clean energy future. The European Union have set a minimum target of 27% of total energy consumption sourced from renewables by 2030 (Delbeke *et al.*, 2016). New Zealand have set a target that 90% of their electricity generation will be derived from renewable sources by 2025 (New Zealand Ministry of Economic Development, 2011). Australia has set a target that 23% of the nation’s electricity be sourced from renewables by 2020 (DoEE, 2015). In addition, Australia has also set an economy-wide target to reduce total emissions by 26-28% below 2005 levels by 2030 (DoE, 2019). Australia has not set a specific GHG reduction target for their agricultural sector, although all sectors of the economy are expected to contribute to this target.

Australia is a small GHG emitting country, at 1.3% of global emissions, compared to other nations, such as China and the USA at 26 and 14%, respectively (CAIT Climate Data Explorer, 2017). However, when compared on a per capita basis, Australia is one of the largest emitters, estimated at 26 t $CO_2$e/head.annum, compared to 9 and 20 t $CO_2$e/head.annum for China and the USA, respectively (CAIT Climate Data Explorer, 2017). Mexico, the UK, South Africa and South Korea have similar percentage of global emissions to Australia, yet have per capita emissions of 6, 8, 10 and 13 t $CO_2$e/head.annum, respectively (CAIT Climate Data Explorer, 2017).

## 8.2 Australia's agricultural greenhouse emissions

The Australian ruminant livestock industry has some of the world's lowest GHG emissions on a per product basis. Gerber *et al.* (2013a) compared the EI of production for beef and small ruminants (sheep and goats) across all regions of the world, including on-farm, post-farm and LUC emissions. Using the Oceania data as representative of Australia, EIs for beef and small ruminants were 25 and 15 kg $CO_2$e/kg carcass weight, respectively, compared to global averages of 45 and 24 kg $CO_2$e/kg carcass weight, respectively. Similar results have been found in other Australian studies (Browne *et al.*, 2011; Wiedemann *et al.*, 2015; Harrison *et al.*, 2016). Gerber *et al.* (2013a) also compared dairy milk production across all regions of the globe and reported an EI of 1.6 kg $CO_2$e/kg FPCM for Oceania, compared to a global average of 2.8 kg $CO_2$e/kg FPCM, with emissions $\geq$ 4.0 kg $CO_2$e/kg FPCM in the developed world. A major contributor to the lower EI of production for Oceania, compared to the global average, is high animal performance (*i.e.* high milk production per cow and fertility along with lower replacement rates) coupled with excellent animal husbandry and welfare practices.

## 8.3 Australian dairy greenhouse emissions

Accounting of Australia's GHG emissions began in 1990 (DoE, 2015), while component mitigation option research began in the early 2000's (Eckard *et al.*, 2003; Grainger *et al.*, 2007, 2009; Kelly *et al.*, 2008). Chapter 3 was the first publication reporting estimates of individual Australian dairy farm GHG emissions, including the pre-farm embedded emissions. The analysis of 60 Tasmanian dairy farms found that annual milk production explained 93% of the difference in annual total farm GHG emissions. The EI of milk production of individual farms varied between 0.8 and 1.4 kg $CO_2$e/kg FPCM, with an overall mean of 1.04 kg $CO_2$e/kg FPCM. This confirms the first hypothesis that the EI of milk production for Tasmanian dairy farms is comparative to other pasture-based systems throughout the developed world. In addition, this analysis was the first time that a SMLR analysis of the three EIs, GHG emissions per kg FPCM, per cow and per hectare, were reviewed against individual key farm variables in Australia. Feed conversion efficiency (kg milk/kg DMI) and N fertiliser (kg N/ha) explained 60% of the difference in EI of milk production across the Tasmanian dataset. This study also found that increasing the proportion of grain in the diet by 10% equated to a 9% increase in FCE, contributing to a reduction in the EI of milk production, thus highlighting grain feeding as a potential mitigation option worth further exploration.

Given the Tasmanian dairy industry makes a small contribution to the nation's milk production (10%; Dairy Australia (2018)), Chapter 4 expanded the GHG emissions analysis to all dairying regions of Australia by reviewing 41 diverse dairy farms, incorporating a range of geographical locations (*e.g.* all dairying regions of the nation) and FSs (*e.g.* incorporation of feed pads). Annual milk production explained 95% of the variation in total farm GHG emissions. The EI of milk production of individual farms varied between 0.8 and 1.7 kg $CO_2$e/kg FPCM, with an overall mean of 1.04 kg $CO_2$e/kg FPCM. This confirms the second hypothesis that the EI of milk production for Australian dairy farms is similar to those in Tasmania and other pasture-based systems throughout the developed world. While the same key farm variables examined in Chapter 3 were repeated in Chapter 4, the SMLR analysis found that milk production per cow (kg FPCM/cow.lactation) could explain 70% of the variation in EI of milk production across the Australian dataset.

Chapter 4 illustrated a link between FS and the EI of milk production. Those farms within FS2 and FS3, reflecting greater grain feeding (*i.e.* > 1 t DM/cow.lactation and without/with confinement for non-grain supplementary feeding, respectively), had a significantly ($P < 0.05$) lower EI of milk production compared to FS1 farms (*i.e.* < 1 t DM/cow.lactation and without confinement feeding of non-grain supplementary feeding). More importantly, FS2 and FS3 farms showed a lower spread of EI between farms; 95% of FS2 and FS3 farms ranged between 0.8 and 1.1 kg $CO_2$e/kg FPCM and 0.9 and 1.1 kg $CO_2$e/kg FPCM, respectively, compared to 95% of FS1 farms varying between 1.0 and 1.6 kg $CO_2$e/kg FPCM. These findings confirm the third hypothesis that FS influences the EI of milk production for Australian dairy farms.

The results from Chapters 3 and 4 illustrated that Australia's EI of milk production is congruent with comparative pasture-based dairying industries from around the world, at approximately 1.0 kg $CO_2$e/kg FPCM. While the average EI of milk production in Australia was comparative to other developed nations, there was a large variation between individual farms, varying between 0.8 and 1.7 kg $CO_2$e/kg FPCM with the national dataset (Chapter 4). This is consistent with the findings throughout the developed world (see Tables 2.1 and 2.2). There is currently no regulatory pressure in Australia to reduce on-farm emissions. However, as we pursue global neutrality of GHG emissions, greater scrutiny will be placed on farms with higher EIs, with an increasingly likelihood that farmers will need to demonstrate their C footprint profile before supplying milk to processors and accessing markets.

As new science emerged in the 2010's, resulting in updates to the NGGI methodology of estimating dairy GHG emissions, the results of Chapter 4 were reviewed using the 2015 updated methodology (Chapter 7). Individual farms varied between 0.8 and 1.5 $CO_2$e/kg FPCM and thus were comparative to the findings in Chapters 3 and 4. There was a substantial increase in the EI of milk production from dairy farms in Western Australia and, to a lesser extent, Victoria. However, for the farms in other regions, such as Tasmania, the state average declined under the updated methodology. Overall, the EI of milk production for the whole Australian dataset increased by 3% to 1.07 kg $CO_2$e/kg FPCM. This finding rejects the fourth hypothesis that changes to the Australian NGGI methodology for estimating dairy farm GHG emissions would result in no difference in the EI of milk production for Australian dairy farms. While difficult to attribute which changes in the updated methodology resulted in specific changes in the EI of milk production for each region, given the mean $N_2O$ emissions from waste and N fertiliser in Tasmania were some of the highest with the old methodology (Chapter 4), the reduction in the EFs and GWP of $N_2O$ in the updated methodology, would have had a greater impact in

Tasmania compared to other states. The increase in EI with Victorian dairy farms is concerning, given that most of Australia's milk is produced in Victoria. The primary contributor to this increase was the change to the integrated MCF, from 6.5% to 9.6% with the old and new methodologies, respectively, due to a larger proportion of waste stored in pond/lagoon systems, at 15% with the new methodology compared to 8% with the previous methodology (DoE, 2015). While the NGGI methodology is a 'one set of rules for all farms' approach to estimate total farm GHG emissions, the need to reduce the proportion of waste collected on 'hard surfaces' is an important focus area for mitigating GHG emissions on all farms.

## 8.4 Greenhouse gas mitigation options

To date, research has yet to identify mitigation options that clearly decouple GHG emissions from milk production. This is a predicted outcome, given the linkages between DMI, milk production and enteric $CH_4$, the largest contributor of on-farm GHG emissions (Hristov *et al.*, 2013a; Knapp *et al.*, 2014; Pickering *et al.*, 2015; Charmley *et al.*, 2016). However, the results from this thesis (Chapters 3, 4 and 7) have highlighted an opportunity to explore why differences in EI between farms occurs and has shown that there is scope within the current FS to adopt technological or management changes that could further reduce the EI of milk production.

Moate *et al.* (2016) reported how changes to the Australian dairy industry between 1980 and 2010 have altered enteric $CH_4$ emissions. On-farm application of research on dairy cattle nutrition and genetic improvements has led to substantial increases in milk production per cow over the last three decades. Cows have become heavier, and with intakes increasing by 40% and milk yields almost doubling, annual enteric $CH_4$ emissions increased by 19% over this time frame. However, this increase in enteric $CH_4$ emissions has been diluted by the substantial increase in milk production per cow (kg milk/cow.lactation), with total enteric $CH_4$ intensity reducing from 33.6 to 23.9 g $CH_4$/kg milk. While milk production per cow was shown to be a key determinant of variability of EI of milk production across the 41 Australian dairy farms (Chapter 4), it is sometimes (Moran *et al.*, 2000; Hanrahan *et al.*, 2018), but not always (Ramsbottom *et al.*, 2015) considered a key driver of farm profitability for pasture-based dairy systems. Therefore, in addition to modest increases in milk production per cow through genetic and feeding management, other improvements will be necessary to continue reducing on-farm enteric $CH_4$ emissions.

While there has been an abundance of research examining management options to reduce enteric $CH_4$ of the dairy cow, there has been minimal focus on the performance of the dairy heifer, a key source of GHG emissions for any dairy farm system. The GHG emissions emitted by dairy heifers from birth to first calving, based on data from Chapter 4, can be as high as 10-20% of total farm GHG emissions. Traditionally, there has been a trend towards a flatter annual milk production curve in northern Australia to supply the domestic liquid milk market. This has facilitated heifers being mated to calve at any time of the year (Hough, 1992; Gilmour *et al.*, 2012; Kempton and Waterman, 2014). Survey data from the northern Australian dairy industry would suggest the age to first calving is approximately 34 months of age, 10 months older than in southern Australia, where heifers are mated to calve at two years of age, to coincide calving with peak seasonal climatic conditions and associated pasture production. This highlights a potential inefficiency in the farm system and thus a source of increased GHG emissions.

Chapter 6 explored the effect of diet quality on the time duration between weaning and first mating for heifers grazing subtropical $C_4$ pastures and cumulative enteric $CH_4$ emissions abatement, using two approaches; (i) static – where daily diet quality was constant and intake matched LWG requirements, and (ii) dynamic – where daily diet intake and quality were variable depending on climate variability. Improving diet quality from 9.5 to 10.9 MJ/kg DM, with the dynamic approach, resulted in heifers reached a target mating LW by 17 months of age on the high-quality diet compared to 22 months on the low-quality diet. Daily enteric $CH_4$ emissions were reduced by 22% and coupled with the 5-month reduction in time between weaning and first mating, cumulative enteric $CH_4$ emissions declined by 42%. When scaled to the whole-of-farm system, total farm net GHG emissions may be reduced by 5 to 10%. While not modelled, assuming the same number of lactations, improving diet quality would also result in a reduction in lifetime EI as the heifer joined the milking herd at an earlier age. The results of Chapter 6 confirm the sixth hypothesis of this thesis that improving the energy density of the diet of heifers would increase their LWG between weaning and first mating, thus reducing cumulative enteric $CH_4$ emissions and EI (kg $CO_2$e/kg LW) over this time period.

Dairy farms are also a source of N loss to the environment, with loss pathways of $N_2O$ denitrification, $NO_3$ leaching and $NH_3$ volatilisation. For grazing-based dairy systems, most of the N loss is associated with urine deposition, with N loading often greater than potential N uptake by pastures or crops (Whitehead, 1995; de Klein and Eckard, 2008). Thus, an effective mechanism to reduce environmental N loss is to lower the amount of N present in urine. Chapter 5 evaluated two management options to improve animal NUE, across three climatic regions of southern Australia. The first management option was to reduce the amount of N consumed in the diet, thus reducing the N source. The second option was to examine the effect of increasing concentration of N in milk, thus increasing the N captured in product. Improving NUE through reducing the overall N concentration of the diet, resulted in greater reductions in $N_2O$ emissions, relative to increasing the N concentration in milk. Nitrous oxide emissions were reduced by between 50 and 57% by reducing the overall N concentration of the diet (pasture and supplement) from 4.1% to between 2.2 and 2.5% across the three regions. Replacing higher N concentrated pasture with a grain or forage supplement (*e.g.* maize silage) with a lower N concentration was shown to be effective in improving NUE and reducing $N_2O$ emissions and at a magnitude greater than increasing N concentration in milk. Thus, the findings of this analysis confirmed the fifth hypothesis of this thesis that a greater reduction in $N_2O$ emissions and the EI of milk production can be achieved by better balancing the energy to N ratio in the milking cow's diet than increasing the N captured in milk. However, given that animal-derived $N_2O$ emissions is a minor component of total farm emissions, this strategy alone may only reduce total farm GHG emissions by up to 10%.

Both mitigation options examined in this thesis were single components of the whole-farm system. While individual sources of GHG emissions were substantially reduced, when scaled to the whole-of-farm system, the result was an abatement of approximately 10%. Similar results have been found when exploring the benefits of feeding dietary fats to milking cows to reduce enteric $CH_4$ emissions. Moate *et al.* (2016) reviewed data from 17 cattle experiments, finding that each additional 1% increase in dietary fat concentration decreased enteric $CH_4$ emissions by 3.5%. On an annual basis, feeding plant-derived fats could reduce enteric $CH_4$ emissions by 10-15%. However, feeding dietary fats is generally restricted to the summer period, when pasture quality can be < 3% fat (Grainger *et al.*, 2008; Legesse *et al.*, 2011). Based on enteric $CH_4$ emissions averaging 55% in Chapters 3 and 4, over a whole year across the whole farm system, the reduction in total farm emissions could be as low as 5% per annum. In addition, the reduction could be partially or completely negated if the GHG emissions associated with the transportation of the dietary fat is taken into consideration (Williams *et al.*, 2014). Therefore, it will be imperative to identify a range of mitigation options that can target different components of the whole farm system, which are additive, synergistic or interactive, are currently possible, and when implemented, reduce the GHG emissions associated with milk production.

### 8.5 A Marginal Abatement Cost Curve analysis of potential mitigation options

While the primary focus of implementing mitigation options may be driven by the need to reduce GHG emissions, the economics of such an implementation also need to be taken into consideration, if adoption by farmers is to be realised. A Marginal Abatement Cost Curve (MACC) analysis is frequently used to compare the marginal cost of abatement against the size of abatement achieved for a range of mitigation options, relative to business-as-usual, thus prioritising alternative mitigation strategies based on their financial characteristics (Bockel *et al.*, 2012; Cotter *et al.*, 2015). Using a MACC analysis, Cotter *et al.* (2015) examined a range of enteric $CH_4$ mitigation options for a typical Australian dairy farm, finding that some options were profitable with a price of AU \$14/t $CO_2$e abated (*e.g.* feeding wheat), while other options remained unprofitable with a price of AU \$50/t $CO_2$e abated (*e.g.* feeding 3NOP).

MACC analyses have also been effectively used to compare mitigation potentials across other agricultural sectors and land management practices (MacLeod *et. al.* 2010; O’Brien *et al.*, 2014c). For example, Moran *et al.* (2011) illustrated that the use of a NI could abate 0.3 t $CO_2$e/ha.annum, equivalent to 604 kt $CO_2$e across the UK agricultural sector by 2022. However, the cost-effectiveness (based on year 2006 prices) was only positive when the value of abatement reached £294/t $CO_2$e, equivalent to approximately AU \$525/t $CO_2$e abatement (based on a current exchange rate of AU \$1.78/£). This high abatement price was a result of the purchased cost of the NI-coated fertiliser being greater than standard N fertilisers with little productivity improvement. As such, the implementation of a NI on farm was not considered financially viable compared to other mitigation options, such as drainage of soils, that could abate 1.0 t $CO_2$e/ha.annum, equivalent to 1,741 kt $CO_2$e across the UK agricultural industry at a cost of approximately AU $25/t $CO_2$e (Moran *et al.*, 2011).

Results from a MACC analysis can highlight profitable mitigation opportunities that can currently be implemented on-farm that are win:win for the environment and the farmer, while also identifying options where either the implementation cost needs to decline, the abatement incentive needs to increase and preferably, a combination of both. Where the implementation of a mitigation option has led to an improvement in productivity (*e.g.* increased milk, meat or fleece production), the additional income generated with the change in production is often an order of magnitude greater than any potential income generated from a market-derived C offset (Doran-Browne *et al.*, 2015; Cottle *et al.*, 2016; Cullen *et al.*, 2016). However, it must be noted that the results of MACC analyses are entirely dependent on the input assumptions. Therefore, these need to clearly be identified and accurately reflect the potential change in GHG emission and cost of implementation.

## 8.6 Exploring farm case studies using a MACC analysis

A MACC analysis was undertaken, reviewing the effect of seven mitigation options across four contrasting Australian case study farms presented in this thesis, taking into consideration the whole farm system and timeframe of activation.

### *8.6.1. Farm selection for MACC analysis*

Four case study farms were selected from the data presented earlier in the thesis, two from FS1 and two from FS2/3. Within each FS, a farm with a high EI (~ $75^{th}$ percentile) and a farm with a low EI (~ $25^{th}$ percentile) were selected so that the combination of FS with EI reflect the variation in EIs across the individual farm results from Chapter 7. Farms were from four different regions of Australia, from south-eastern and northern Victoria, south and north coast of New South Wales, thus incorporating geographic diversity. Farms were reviewed to confirm that the mitigation options selected could be implemented across all four farms (*e.g.* adequate N fertiliser inputs to assess the effect of a NI on reducing $N_2O$ emissions). Table 8.1 contains key farm data for each of the four farms that are relevant to the MACC analysis. These four farms were re-entered into the ADCC (Christie *et al.*, 2012) to estimate their baseline farm GHG emissions and EI of milk production to allow further analysis of each mitigation option.

**Table 8.1** Key farm data for the four farms subjected to the MACC analysis to ascertain the effect of seven mitigation strategies to reduce greenhouse gas emissions.

| | **FS1 High EI** | **FS1 Low EI** | **FS2/3 High EI** | **FS2/3 Low EI** |
|---|---|---|---|---|
| Farm location | NC NSW | SE VIC | SC NSW | Nth VIC |
| Milking herd size (number of milkers) | 300 | 231 | 290 | 710 |
| Replacement herd size[1] | 56 | 46 | 70 | 190 |
| Replacement rate (%) | 19 | 20 | 24 | 27 |
| Milking platform area (ha) | 171 | 93 | 221 | 236 |
| N fertiliser (‘000 kg N) | 14.4 | 19.1 | 30.0 | 15.8 |
| N fertiliser (kg/ha) | 72 | 140 | 117 | 17 |
| Concentrate intake (t DM/milker) | 0.89 | 0.71 | 2.85 | 1.42 |
| Dietary DMD (%) | 75.7 | 75.0 | 76.1 | 76.5 |
| Dietary CP (%) | 19.5 | 21.8 | 21.5 | 17.8 |
| Dietary fat (g/kg DM) | 3.6 | 4.1 | 3.1 | 3.2 |
| Milk production (kg FPCM/cow) | 4,312 | 5,322 | 7,671 | 7,032 |
| Baseline farm total GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline farm EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |

[1] Same number of rising 1 year olds as rising 2 year olds, with this figure referring to the number of stock in each age class

CP, crude protein; DM, dry matter; DMD, dry matter digestibility; EI, emissions intensity; FPCM, fat and protein-corrected milk; FS, farming system; GHG, greenhouse gas; NC NSW, north coast New South Wales; SC NSW, south coast new South Wales; SE VIC, south eastern Victoria; Nth VIC, northern Victoria.

### *8.6.2. The use of the Carbon Offset Scenario Tool for estimating change in GHG emissions*

Christie *et al.* (2013) developed the empirical spreadsheet model Carbon Offset Scenario Tool (COST), which is embedded within the ADCC. A baseline farm data is entered and then the COST provides nine mitigation options across the four broad theme areas to be explored; (i) diet manipulation, (ii) herd management, (iii) breeding management, and (iv) feedbase management. In addition, a full replication of the baseline farm allows the user to alter individual components of the baseline farm (*e.g.* alter herd numbers, energy consumption, tree plantings for C sequestration etc). Key variables for each mitigation option, identified from prior research, and a price for milk and C offsets can be adjusted, allowing users to quickly explore the impact of the variables on farm GHG emissions and profitability. Results are presented as changes in C offset income, mitigation option implementation cost, change in milk production income and net farm income on a per annum and a per EI of milk production basis (Christie *et al.*, 2013).

### *8.6.3. Mitigation strategies explored in COST*

For this analysis, seven mitigation options were explored in COST that reflected management practices that farmers could currently implement and could be considered win:win strategies for reducing total farm GHG emissions while maintaining or improving productivity and profitability. These mitigation options were:

(a) genetic improvement, across all stock classes, to reduce the proportion of energy lost as enteric methane ($CH_4$) production (herein referred to as **Genetic improvement**);

(b) feeding a source of dietary fats to the milking herd to reduce enteric $CH_4$ production (herein referred to as **Feeding fats**);

(c) feeding a low protein supplement to the milking herd to improve the energy to protein ratio of the diet of the milking cow to reduce urinary $N_2O$ emissions (herein referred to as **Improved DMD to CP ratio**);

(d) use of a NI- coated fertiliser to reduce soil $N_2O$ emissions (herein referred to as **NI- N fertiliser**);

(e) use of a NI sprayed onto urine patches of the milking cow (whole paddock) to reduce urinary $N_2O$ emissions (herein referred to as **NI- urinary deposition**);

(f) reducing the cow replacement rate to reduce enteric $CH_4$ and animal $N_2O$ emissions (herein referred to as **Reduced RR**);

(g) extended lactation of the milking herd to reduce enteric $CH_4$ and animal $N_2O$ emissions (herein referred to as **Extended lactation**).

The Improved DMD to CP ratio mitigation option was selected to reflect the modelling undertaken in Chapter 5, where reducing the SN concentration of the diet improved the DMD to CP ratio of the diet of the milking cow to reduce $N_2O$ emissions. The Reduced RR mitigation option was selected to reflect an aspect of the modelling undertaken in Chapter 6. The research in Chapter 6 reviewed the effect of improved diet DMD to reduce time to first mating and cumulative enteric $CH_4$ emissions. It did not consider that fewer replacement animals would be required to maintain a similar replacement rate each year. In the original analysis of farms in Chapter 7 to estimate baseline EIs, it was assumed that the DMD of the replacement animals' diet was already high at 75%, and that the rising two years old heifers were assumed to have calved at two years of age. Thus, for this MACC analysis, it was not possible to change diet quality or age of calving, compared to the baseline farm. Therefore, only the effect of reduced replacement rate on reducing $CH_4$ and $N_2O$ emissions from these animals was explored in this analysis.

Each of the seven mitigation options was applied, in isolation, to each of the four farms to ascertain the effect of each mitigation option on EI of milk production and total farm GHG emissions. A series of assumptions were made for all farms across all mitigation options (*e.g.* same price for any additional milk produced). In addition, each mitigation option also had a series of assumptions that were consistent across each farm, although the result of the assumption could vary between farms. For example, with the Improved DMD to CP ratio, the assumption was to replace half of the grain in the diet with maize silage, with the amount of grain replaced with maize silage varying between farms. See Appendix 2 for the series of assumptions implemented across all farms and mitigation strategies and Appendix 3 for an explanation of the assumptions for each mitigation option across all four farms.

For four of the seven mitigation options (Feeding fats, Improved DMD to CP ratio, NI- N fertiliser and NI- urinary deposition) the assumption was that these mitigation options could be implemented immediately (within the current year). However, for the other three mitigation options (Genetic improvement, Reduced RR and Extended lactation), full implementation of the mitigation option would take several years to accomplish. For example, to implement the Genetic improvement mitigation option, the whole herd (milkers and replacement animals) requires every higher emitting animal to be replaced with a lower emitting animal, through years of selective breeding. The assumption for this MACC analysis was that these three mitigation options would require six years of transition to achieve full implementation. See Appendix 3 for an explanation of the pattern of transition for these mitigation options.

As each farm varied in size and thus the baseline total farm GHG emissions varied, it was necessary to compare farms on a per kg FPCM basis. The reduction in total farm GHG emissions for each mitigation option was estimated as the difference in EI between the baseline farm and the strategy farm (post-implementation) multiplied by the strategy farm's milk production. This reduction in total farm GHG emissions was then divided by the net cost of implementation (annual cost to implement minus any annual change in milk production income) to determine the cost-effectiveness (AU \$/t $CO_2$e abatement). For example, implementation of a mitigation option reduced EI of a baseline farm from 1.0 to 0.9 kg $CO_2$e/kg FPCM, with milk production increasing to 500,000 kg FPCM. The mitigation option reduced EI by 0.1 kg $CO_2$e/kg FPCM and abated 50,000 kg $CO_2$e (500,000 kg FPCM x 0.1 kg $CO_2$e/kg FPCM). If the cost of implementation was -\$1,000/annum, the cost-effectiveness was estimated as -\$20/t $CO_2$e abated (-\$1,000 / 50 t $CO_2$e abated).

The cost of implementation was assumed to remain the same each year for the four immediately implemented mitigation options (*e.g.* no change in grain or maize silage costs). For the three mitigation options that took six years to implement, the net implementation cost was estimated on an annualised basis using a Net Present Value function with 5% discount rate in COST such that all seven mitigation options could be compared equally on an annualised basis.

A mitigation option that resulted in a decrease in the cost of implementation (*e.g.* reduced supplementary feed costs or increased income generated from increased milk production) is denoted by a negative cost of abatement (*e.g.* -$10/t $CO_2$e abated). When coupled with a reduction in GHG emissions, this represents a win:win mitigation option in terms of reducing GHG emissions while maintaining or increasing productivity and/or profitability. Therefore, this mitigation option would be profitable to implement in the absence of any potential income generated as part of a C market. In contrast, a mitigation option that resulted in an increase in the cost of implementation (*e.g.* additional purchased supplementary feed or decrease in income generated from decreased milk production) is denoted by a positive cost of abatement (*e.g.* +$10/t $CO_2$e abated). While this mitigation option resulted in a reduction in GHG emissions combined with a potential increase in productivity, the option is not financially profitable to implement in its current state. A price premium for the lower GHG emission milk, generated as part of a C market, would decrease the cost of abatement, thus potentially changing this mitigation option from being cost-restrictive to being income-generating.

### *8.6.4. Results of the MACC analysis*

All mitigation options resulted in a decline in EI, relative to the corresponding baseline farm EI, thus when this difference in EI was multiplied by milk production of the strategy farm, total farm GHG emissions declined (Table A4a to A4f). The only exception to this was with the Improved DMD to CP ratio mitigation option when implemented on the FS2/3 Low EI farm and when the silage was fed in the paddock. The EI of milk production increased by 0.003 kg $CO_2$e/kg FPCM (Figure 8.1d), thus total farm GHG emissions increased by 13 t $CO_2$e/annum (Table A4c). However, when the silage was assumed to be fed on a feedpad, as opposed to in the paddock, EI and total farm GHG emissions declined, relative to the baseline farm, and thus became an effective mitigation option to implement (Table A4c in brackets).

Figure 8.1 illustrates the marginal abatement cost ($/t $CO_2$e abated) and abatement potential (kg $CO_2$e/kg FPCM) for the seven mitigation options across the four contrasting farms.

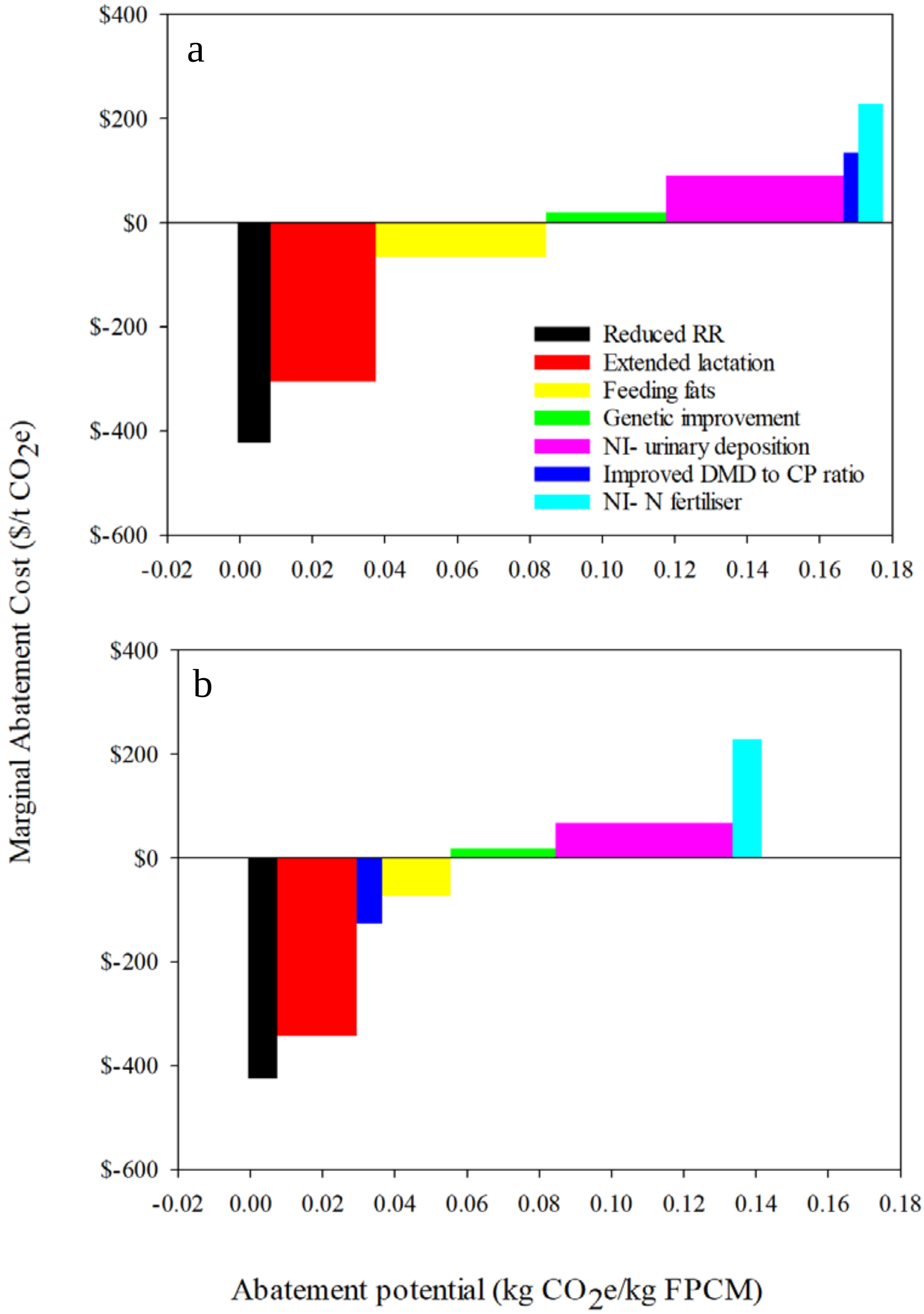


**Figure 8.1** Annualised greenhouse gas abatement potential (kg $CO_2e$/kg FPCM) and marginal abatement cost ($/t $CO_2e$) for a range of mitigation options implemented on a FS1 High EI farm (a) and FS1 Low EI farm (b).

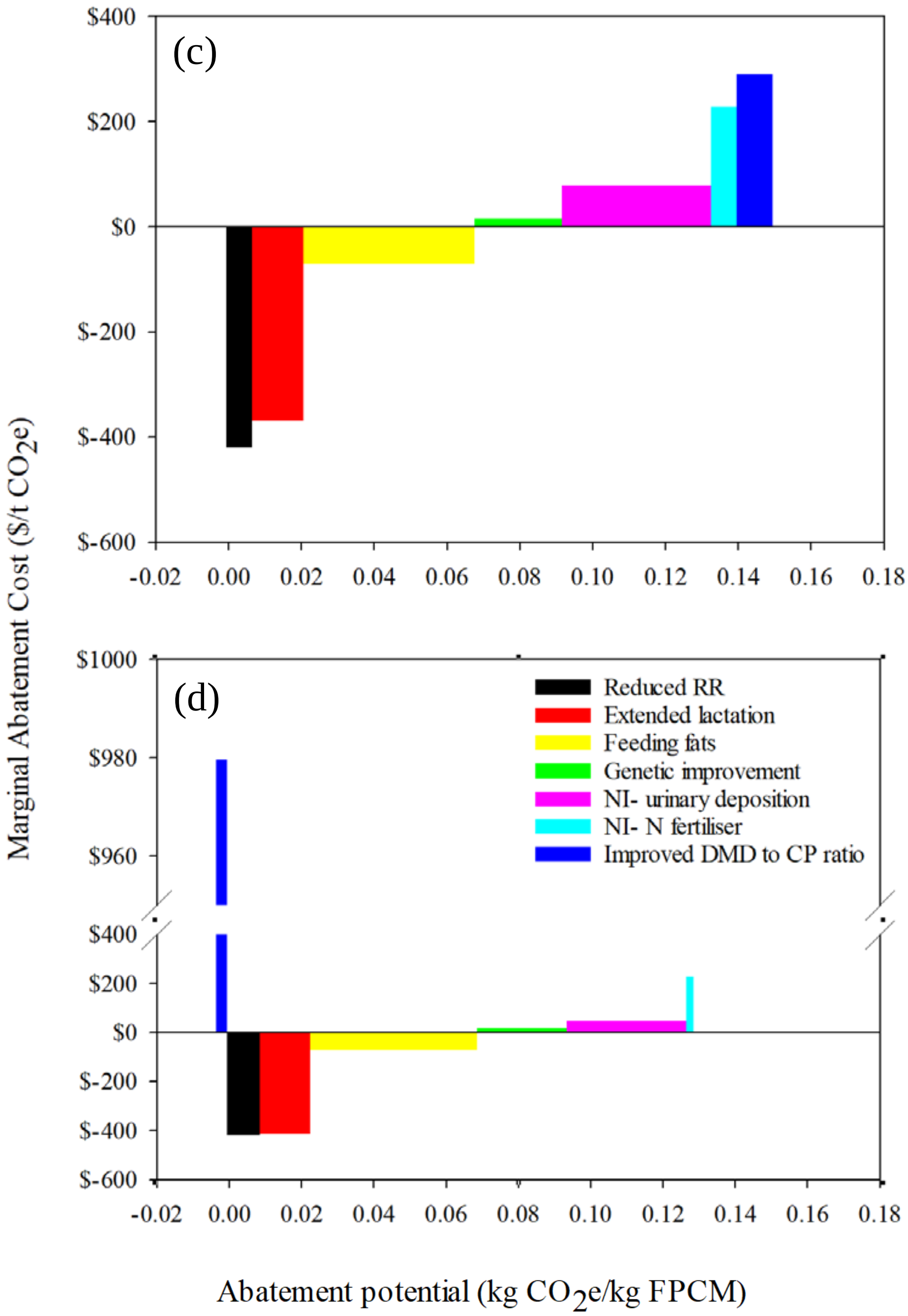


**Figure 8.1 cont.** Annualised greenhouse gas abatement potential (kg $CO_2$e/kg FPCM) and marginal abatement cost ($/t $CO_2$e) for a range of mitigation options implemented on a FS2/3 High EI farm (c) and FS2/3 Low EI farm (d). NOTE the scaling differences between the (d) graph and the other three graphs due to a high marginal abatement cost for the Improved DMD to CP ratio mitigation option. In addition, this mitigation resulted in a negative abatement potential for the FS2/3 with low EI farm.

#### 8.6.4.1. *Individual mitigation option results*

The Genetic improvement mitigation option resulted in a decline in EI of between 0.024 and 0.033 kg $CO_2$e/kg FPCM, thus annualised total farm GHG emissions reduced by between 35 and 128 t $CO_2$e/annum (Table A4a). This mitigation option cost between +$16 and +$20/t $CO_2$e abated, thus while a win for the environment, it was not a profitable option to implement (Figure 8.1).

Feeding fats resulted in a decline in EI of between 0.019 and 0.048 kg $CO_2$e/kg FPCM, thus total farm GHG emissions reduced by between 24 and 233 t $CO_2$e/annum. This mitigation option resulted in a profit of between $65 and $74/t $CO_2$e abated, thus a win:win for the environment and farm profitability (Figure 8.1).

The Improved DMD to CP ratio mitigation option resulted in decline in EI of 0.004 kg $CO_2$e/kg FPCM for the FS1 high EI farm, increasing to a decline of 0.007 for both the FS1 Low EI farm and the FS2/3 High EI farm. Total farm GHG emissions declined by 5, 9 and 15 t $CO_2$e/annum for the FS1 High EI, FS1 Low EI and FS2/3 High EI farm, respectively (Table A4c). In contrast, EI increased by 0.003 kg $CO_2$e/kg FPCM for the FS2/3 Low EI farm (Table A4c). Total farm GHG emissions increased by 13 t $CO_2$e/annum and thus was not a win:win mitigation option for this particular farm. While the Improved DMD to CP ratio mitigation option was profitable for the FS1 low EI farm, at -$125/t $CO_2$e abated (Figure 8.1b), it was unprofitable for the other three farms, costing +$135 and +$415/t $CO_2$e abated for the FS1 High EI farm (Figure 8.1a) and FS2/3 High EI farm (Figure 8.1c), respective, and costing +$980 for each t $CO_2$ emitted with the FS2/3 Low EI farm (Figure 8.1d). The above-mentioned results for the two FS2/3 farms assumed the silage was fed in the paddock, assuming 30% wastage. When the silage was fed on a feedpad, with only 10% wastage, relative to the baseline farm, EI was reduced by 0.016 and 0.002 kg $CO_2$e/kg FPCM for the High EI and Low EI farm, respectively (results in brackets in Table A4.c). Total farm GHG emissions reduced by 36 and 12 t $CO_2$e/annum, and was profitable at -$329 and -$761/t $CO_2$e abated for the Low EI and High EI farm, respectively (results in brackets in Table A4c). These differences highlight the importance of assumptions in estimating and interpreting results, especially for the FS2/3 Low EI farm transitioning from an increase in EI to a decrease in EI.

The NI- N fertiliser mitigation option resulted in a decline in EI of between 0.002 and 0.008 kg $CO_2$e/kg FPCM and total farm GHG emissions reduced by between 8 and 16 t $CO_2$e/annum (Table A4d). The cost of implementation was consistent across all four farms, at +$228/t $CO_2$e abated, due to the same magnitude of reduction of N2O emissions per kg fertiliser and the cost of the NI based on a $/t N fertiliser rate (Figure 8.1).

The NI-urinary deposition mitigation option reduced EI by between 0.033 and 0.050 kg $CO_2$e/kg FPCM, thus total farm GHG emissions declined by between 60 and 166 t $CO_2$e/annum (Table A4.e). Applying a NI to decrease N losses from urine deposition cost between +$45 and +$92/t $CO_2$e abated (Figure 8.1). When the FS2/3 cows spent an additional 5% of their time on a feedpad, EI and total farm GHG emissions both increased slightly, although still lower than for the baseline farm. The cost of implementation increased by $3 to $5/t $CO_2$e abated, to be +$48 and +$84/t $CO_2$e abated for the Low EI and High EI farm, respectively (results in brackets in Table A4e).

The Reducing RR mitigation option reduced EI by between 0.006 to 0.008 kg $CO_2$e/kg FPCM, with annualised total farm GHG emissions declining by between 9 and 38 t $CO_2$e/annum (Table A4f). This mitigation option was profitable, varying between -$419 and -$425/t $CO_2$e abated (Figure 8.1), thus a win:win for the environment and farm profitability.

The Extended lactation mitigation option incurred additional pre-farm embedded GHG emissions in addition to an increase in on-farm GHG emissions associated with an increase in annualised milk production. However, the EI of milk production declined by between 0.014 and 0.028 kg $CO_2$e/kg FPCM as a result of increased milk production. Thus, annualised total farm GHG emissions declined by between 15 and 32 t $CO_2$e/annum (Table A4g), was profitable by between -$305 and -$413/t $CO_2$e abated (Figure 8.1) and thus a win:win option for the environment and farm profitability.

#### *8.6.4.2. Comparison of results across and within farming systems*

The order of mitigation options in terms of cost-effectiveness was consistent across all four farms, except for the Improved DMD to CP ratio mitigation option (Figure 8.1), mostly likely due to the static nature of COST and consistency of implementation of assumptions across all four farms. Thus, the MACC analysis was unable to formulate a clear variation in ranking of mitigation options either between or within FS in terms of cost-effectiveness. However, there were some clear differences in the abatement potential between farms, highlighting the need for reviewing farms individually to ascertain the suitability of implementing mitigation options. This was most evident with the Improved DMD to CP ratio mitigation option, where the cost-effectiveness was contingent on the baseline diet quality as opposed to the FS or EI of milk production. This option was only profitable for one of the four farms (FS1 Low EI farm; Figure 8.1b), in addition to increasing the EI of milk production for another of the four farms (FS2/3 Low EI farm; Figure 8.1d). Therefore, understanding aspects of the baseline farm where improvements can be achieved is more critical to deciding if a mitigation option will be environmentally beneficial while maintaining or improving farm productivity and profitability than a uniform recommendation based on FS.

Within a FS, the higher EI farm had more scope to reduce GHG emissions compared to the lower EI farm. Much of this may be due to the static nature of COST in addition to the assumptions for these mitigation options. This is a clear limitation of undertaking a GHG mitigation analysis this way and highlights the need for dynamic modelling analyses and/or field experimentation to confirm the results. The NI- N fertiliser mitigation option was a good example of how a static approach to estimating the reduction of $N_2O$ emissions did not take all the contributing factors into consideration (*e.g.* soil moisture and temperature, rate and timing of application) and thus, there was no difference in the cost-effectiveness between and within FSs for this mitigation option.

### 8.6.5. *Discussion of the MACC analysis*

Mitigation options that deliver a reduction in enteric $CH_4$ emissions, animal waste $N_2O$ emissions, N fertiliser $N_2O$ emissions and/or pre-farm embedded emissions, while producing the same or more product, ideally from fewer animals, will result in both a reduction in total farm emissions and EI and as such should be the goal for farmers to achieve. It must be noted that the results achieved with this MACC analysis was a direct result of the input assumptions and different input assumptions would alter the GHG reduction potential and cost-effectiveness of each mitigation option.

#### 8.6.5.1. *Genetic improvement*

The Genetic improvement mitigation option was marginally unprofitable to implement, costing between +$16 and +$20/t $CO_2$e abated. Farmers could decide to absorb this cost within the farm budget or the cost for achieving the genetic improvement could reduce. Ideally, the farmer receives a price premium for producing milk with a lower emissions footprint with no additional cost to implement. The mode of delivery in reducing enteric $CH_4$ emissions was not identified in this analysis, only a 10% reduction potential. Breeding for residual feed intake has been shown to be effective in reducing enteric $CH_4$ emissions (Waghorn and Hegarty, 2011; Connor *et al.*, 2013). Australian dairy farmers can already select semen from bulls with greater weighting placed on improved feed efficiency (Australian Dairy Herd Improvement Scheme, 2015). However, poor repeatability and high within-animal variation between residual feed intake and $CH_4$ emissions has been found (Pinares-Patiño *et al.*, 2007; de Haas *et al.*, 2011; Lassen and Løvendahl, 2016). Other options that could deliver a genetic GHG emission reduction include breeding for increased passage rate (Pinares-Patiño *et al.*, 2011; Huhtanen *et al.*, 2016) or improved FCE (Clark *et al.* (2005; Danielsson *et al.*, 2017).

While the assumptions for this mitigation option were justifiable, caution is needed when considering the effect of these assumptions on overall results, especially with respect to the cost of implementation and the rate of transition to a lower emitting herd. The increased emphasis on selective breeding to deliver reduced GHG emissions may reduce the genetic diversity of the herd (MacLeod *et al.*, 2015), thus reducing potential milk production gain over time and thus the profitability of this option.

### *8.6.5.2. Feeding fats*

Feeding fats reduced net enteric $CH_4$ emissions and increased milk production during the period of activation, thus both contributed to the decline in EI. If this analysis had assumed that the energy content of the high fat supplement was the same as the grain it was replacing, there would have been no change in milk production and thus erode some of the profitability when implemented. In addition, this analysis assumed that the high-fat supplement cost only 20% more than the grain it replaced. This is in contrast to some other studies (MacLeod *et al.*, 2015) where feeding fats was assessed as being cost-prohibitive due to a greater increase in the cost of the high-fat supplement diet, relative to the baseline lower fat content diet, coupled with no animal productivity increase (Henderson *et al.* 2017). It is plausible that as farmers seek options to reduce on-farm GHG emissions, the cost differentiation between grain and high fat supplements may increase due to changes in market demand and thus reduce the cost-effectiveness of this mitigation option. This was especially the case during 2018/19 due to drought conditions across most of the country resulting in escalated prices for all supplementary feeds. Grain prices were $450/t DM, with high fat supplements significantly greater than this, thus decreasing the financial viability of purchasing grain, let alone high fat supplement, for many farmers.

While feeding dietary fats was financially and environmentally profitable, based on the MACC analysis assumptions, Williams *et al.* (2014) and Ludemann *et al.* (2016) found that transportation distance of the supplement would need to have a very low/zero GHG burden, to maintain the GHG-effectiveness of this mitigation option. This MACC analysis assumed no transportation GHG emissions were included in the analysis, on the assumption that one supplement merely replaced the other, resulting in no net change in transport emissions. In addition, this analysis assumed no change in milk fat composition, and thus milk income, with feeding a high fat supplement. Some researchers have found that feeding a high fat supplement can result in a depression in milk fat concentration (Beauchemin *et al.*, 2008; Moate *et al.* 2014), although this has not always been observed (Eugène *et al.*, 2008; Moate *et al.* 2011). Thus, any change in the assumptions for this mitigation option could have resulted in a different outcome in terms of EI of milk production, total GHG emissions reduction and cost-effectiveness.

### *8.6.5.3. Improved DMD to CP ratio*

The Improved DMD to CP ratio mitigation option was only profitable for the FS1 Low EI farm. While this farm had the lowest rates of grain feeding of all four farms, the baseline diet had the highest CP concentration, at 21.8% and thus scope for being reduced closer to the ideal 16-18% for milking cows (Whitehead, 1995). In addition, the diet DMD concentration for this farm was practically the same as the grain and maize silage supplement, thus only decreasing total herd milk production by 0.2 t FPCM/annum, compared to between 5.6 and 43.9 t FPCM/annum for the other three farms (Table A4.c).

In contrast, while improving the DMD to CP ratio of the diet reduced total farm GHG emissions for the two high EI farms, the decline in milk production resulted in an increase in the cost of implementation, at +$135 and +$415/t $CO_2$e abated for the FS1 and FS2/3 farm, respectively. As such, it could be concluded that this option was not a win:win mitigation option for these two high EI farms, as the reduction in milk production, and thus farm profitability, was substantially greater than the reduction in total farm GHG emissions.

Improving the DMD to CP ratio mitigation option was also not suitable to implement for the FS2/3 Low EI farm, as EI and total farm GHG emissions increased, relative to the baseline farm, primarily driven by this farm already having a high DMD to CP ratio with an overall diet DMD of 76.5% and CP of 17.8% (Table 8.1). Milk production was reduced due to the feeding of 75% DMD grain and silage supplement, in addition to only a small reduction in waste $N_2O$ emissions compared to the other three farms.

Two rates of silage wastage were examined for the two FS2/3 farms; a 30% wastage rate, indicative of a FS2 farm where the silage was fed on the paddock and a 10% wastage rate indicative of a FS3 farm where the silage was fed on a feedpad. It was critical when examining the effect of the silage wastage rate for each farm, that the herd was maintained on the same level of DMI and milk production, so that on-farm GHG emissions did not alter. Each 1kg DM of grain was replaced with either 1.3 kg DM of silage for the FS2 farm or 1.1 kg DM of silage for the FS3 farm, altering the pre-farm gate embedded emissions and cost of implementation. The results presented in Figure 8.1 were of the FS2 farm where 30% of the silage was wasted in the paddock. This mitigation option was not cost-effective for either the high or low

FS2/3 farms when silage was fed in the paddock. However, when the silage was fed on a feedpad, the mitigation option became cost-effective for both farms, due to a decline with both the pre-farm gate embedded emissions reducing EI and total farm GHG emissions, in addition to the reduction in cost of purchasing less maize silage. This highlights the importance of determining appropriate assumptions in addition to the benefit of reducing wastage of supplementary feeding in further reducing total farm GHG emissions.

In this MACC analysis, the amount of maize silage introduced into the diet was low (< 13% of the diet intake for three of the four farms and 23% for the fourth farm (FS2/3 High EI)). This is in contrast to the results in Chapter 5, where between 40 and 56% of the diet was from supplementary feeding with varying rates of N concentration. There is a need to identify and implement management strategies that better balance the DMD to CP ratio of the largest component of the diet for the majority of Australia's dairy industry, grazed pasture, as opposed to replacing the generally smaller component of the diet, grain, to achieve greater reductions in GHG emissions. If this can occur on-farm, through management practices that do not incur any additional cost of implementation, such as pasture species selection or better grazing management (Rawnsley *et al.*, 2002; Turner *et al.*, 2006; Ludemann *et al.*, 2015; Lawson *et al.*, 2017), this mitigation option could further improve the reduction in total GHG emissions and EI of milk production while also becoming profitable as opposed to cost-restrictive.

#### *8.6.5.4. NI- N fertiliser*

Coating N fertiliser with a NI was less effective in reducing on-farm GHG emissions compared to spraying the NI onto pastures post-grazing. This was primarily driven by GHG emissions from N fertiliser (direct and indirect $N_2O$ emissions) only being between 1.0 and 4.3% of total farm GHG emissions (data not shown), thus little scope to reduce total farm emissions. The EI of milk production only declined by between 0.002 and 0.008 kg $CO_2$e/kg FPCM; one of the smallest reductions in EI across the seven mitigation options. With an application cost of +\$228/t $CO_2$e abated (same for all four farms as calculated on a \$/t N fertiliser basis), this mitigation option would be considered unviable for farmers to currently implement. In addition, this cost of implementation would be exacerbated if no additional pasture production (Dougherty *et al.*, 2016) was realised that could be converted into additional milk production, to further reduce EI, or to decrease the reliance on purchased supplements, to reduce input costs.

The abatement potential of this mitigation option was directly linked to N fertiliser rates, with rates varying between 17 kg and 140 kg N/ha.annum across the four farms (Table 8.1). Using a static approach to estimate the reduction in $N_2O$ emissions does not take into consideration the range of factors that influence $N_2O$ loss, such as soil type, climatic conditions and timing/management of N fertiliser application, as seen in field experimentation (Kelly *et al.*, 2008; Suter *et al.*, 2016). This affirms the need for continued field experimentation in addition to more dynamic modelling across a range of locations with varying soil types and climatic conditions to better capture the abatement potential of this mitigation option.

#### *8.6.5.5. NI- urinary deposition*

The NI-urinary deposition mitigation option reduced the EI of milk production by between 0.033 and 0.050 kg $CO_2$e/kg FPCM, thus reducing total farm GHG emissions by between 60 and 166 t $CO_2$e/annum when the cows were all assumed to be grazing pastures for 80% of the year, with minor differences when the FS2/3 farm cows spent an additional 5% of their time on a feedpad. This mitigation option resulted in the largest abatement potential for the two FS1 farms and second largest for the two FS2/3 farms. Therefore, there is merit in further comparing these modelling results with field experimentation to confirm the GHG reduction potential, as the inhibitor was assumed to be effective for half of the year, irrespective of climatic and soil conditions being conducive to $N_2O$ loss. The cost of this mitigation option varied between +$48 and +$92/t $CO_2$e abatement (with only minor adjustments when cows spent longer on a feedpad for the FS2/3 farms) as a function of the implementation cost calculation being on a $/ha basis. This would also need further analysis to confirm the cost of implementation as a lower cost of implementation would need to be realised to make this option more viable.

#### *8.6.5.6. Reducing RR*

The Reducing RR mitigation option would appear to be a very suitable mitigation option for uptake, reducing EI and total farm GHG emissions while remaining profitable for the farmer. However, there are some broader considerations that a MACC analysis does not capture. Reducing the number of replacements will delay genetic gain of positive animal traits (*e.g.* increased milk production, improved feet and udder health) as the overall age of the herd is older. Farmers will have less opportunities to cull older cows based on lower milk production or health constraints, both potentially increasing the EI of milk production for the whole herd. Peak lactation yield also generally occurs in the third to fifth lactation, so retaining cows longer than this results in lower production per lactation (Ray *et al.*, 1992; Vijayakumar *et al.* 2017). If these replacement animals are not being retained on the farm of birth to enter the milking herd, there is a high likelihood that they may be purchased by other dairy farmers, either nationally or internationally, thus shifting their GHG contribution elsewhere. If not grown out to enter a milking herd, then the calves may be euthanised soon after birth and this has social and ethical concerns for the dairy industry. Therefore, the results of a MACC analysis alone can be insufficient as to why a mitigation option should or should not be implemented on-farm.

#### *8.6.5.7. Extended lactation*

Across all four farms, total farm GHG emissions increased, relative to their baseline farm emissions (Table A4g). This contrasts with all other mitigation options examined where total GHG emissions of the strategy farm were lower than those of the baseline farm. Therefore, on first assessment, it could be assumed that this was not a suitable mitigation option as it resulted in increased absolute total farm GHG emissions. However, the Extended lactation mitigation option resulted in an increase in annualised milk production, thus diluting the increase in total farm GHG emissions and consequently, EI declined. Calculating the reduction in total farm GHG emissions as the product of this decline in EI by the increase in milk production generated, total farm GHG emissions declined. Given the increase in milk income was greater than the cost for the additional purchased supplementary feed to support the extended lactation, this mitigation option was the second most profitable to implement.

This mitigation option, more than the other six examined here, highlighted the importance of how total farm GHG emissions are estimated. The Australian beef industry, with the Emissions Reduction Fund (ERF) Beef Cattle Herd Management methodology (see Section 8.10 later in the thesis for an explanation of the ERF), promotes the crediting of a reduction in EI as a practical way of supporting and rewarding economic growth within the industry (DoEE, 2019). A similar methodology for the Australian dairy industry would allow farmers to be credited when implementing extended lactations across their milking herd if there was a reduction in EI achieved with the extended lactation.

Some studies have indicated that extended lactation means the number of replacement heifers produced annually is fewer and thus the number of replacements required also decreases (van Amburgh *et al.*, 1997; Borman *et al.*, 2004; Browne *et al.*, 2015). This would also be the case if maintaining the same annualised milk production from fewer milking cows (Wall *et al.*, 2012). In this MACC analysis, it was assumed that there would be no change in the number of replacement animals required. One-sixth of the herd was replaced annually with the traditional 300-day lactation while one-fourth of the herd was replaced every 18 months with the 482-day extended lactation. Thus, over the six-year period, the same number of heifers would be retained as replacement animals, irrespective of lactation system examined. The results of this analysis contrasted with those of others, such as Browne *et al.* (2015), where increased milk production per cow was coupled with a small reduction in replacement rate, resulting in a substantially greater reduction in total farm GHG emissions and EI of milk production. As discussed previously, reducing the replacement rate was a profitable mitigation, but with far-reaching implications such as the decline in genetic improvement across the whole herd. Thus, it is important to understand the assumptions and their effect on the emissions profile when comparing results of MACC analyses.

In addition, there were many other costs that were not considered as part of this analysis *e.g.* milking shed chemicals and rubberwear, additional laneway maintenance, reproduction costs, calf rearing costs etc. Thus, a more detailed financial analysis of this mitigation option would be required to verify the profitability of altering the calving interval from a standard 12-month to an extended 18-month period.

## 8.7 Combining mitigation options

Each of the seven mitigation options were modelled in isolation and thus the sum of the abatement potentials for each farm in Figure 8.1 does not represent the cumulative sum of combining each option together. The GHG reduction potential of mitigation options that target completely different GHG emission pathways could be summed together. For example, the Feeding fats and NI- N fertiliser mitigation options target two distinct pathways. Thus, the reduction potential of each option could be summed together to estimate their cumulative effect. In contrast, Improving the DMD to CP ratio and NI-urinary deposition mitigation options both target a reduction in animal waste $N_2O$ emissions. Reducing the CP concentration of the diet would reduce urinary N output. Thus summing the abatement potential of each together would over-estimate the abatement potential of combining these mitigation options.

There is published research where authors have examined a combination of mitigation options to reduce dairy GHG emissions. For example, del Prado *et al.* (2010) modelled the scope to mitigate GHG emissions for a typical UK dairy system by implementing changes to management (fertilisation, diet and system changes) and genetics (new animal and plant traits). Individual strategies reduced EI ($CO_2$e/litre milk) by between 1% and 14%. When eight mitigation options were modelled in combination, the EI of milk production was reduced by 45%, lower than the aggregation of each mitigation strategy reduction, at approximately 60%, indicating interaction between two or more strategies.

Another example of the cumulative effect of combining mitigation options is the research of Beukes *et al.* (2017) where they examined the effect of five mitigation options (N fertiliser reduction, improved genetics and reproductive performance, low protein supplementation and use of restricted grazing) using a modelling approach, with the options implemented in field experimentation over four years. Modelling was undertaken for the Waikato region of New Zealand, exploring three different individual years, representing a wet, average and dry year. Modelling the implementation of the five options reduced N leaching by between 31% and 47% for the dry and wet years, respectively. When the same five options were implemented in the field, N leaching was reduced by an average of 43% over the four years, thus within the range found with modelling. Modelling provides an effective and time-efficient way to explore the interactive and additive effects of a range of mitigation options targeting varying facets of the farm system to reduce farm GHG emissions.

### 8.8 Considering implications of mitigation options

All FS1 farms in Chapter 4, indicating low levels of grain feeding, had an EI > 1 kg $CO_2$e/kg FPCM, suggesting a link between grain feeding and EI. Therefore, increasing grain feeding could be a practical win:win mitigation practice for all dairy farmers to adopt to reduce the EI of milk production. However, the foundation and market-advantage of Australia's dairy industry is the grazing of pastures to meet most of the herd's daily intake requirements. Maximising milk production per cow through greater grain feeding at the expense of grazed pastures and breeding larger animals, with the possibility of increasing the reliance on partial or full confinement feeding to capitalise on the additional grain feeding, would most likely decrease the competitiveness of the industry in the global marketplace (Beca, 2005; Savage and Lewis, 2005; Chapman *et al.*, 2008a). It is also important to note that confinement feeding does not necessarily result in a reduction in the EI of milk production (O'Brien *et al.*, 2012; Arnott *et al.*, 2015; Lorenz *et al.*, 2019), especially when pasture-based systems can maintain or increase soil C sequestration (Belflower *et al.*, 2012).

Farm system intensification (FS1 → FS5) through increased grain feeding does not necessarily lead to higher milk production per cow (kg FPCM/cow.lactation) and thus contribute to a lower EI, as was demonstrated in Chapter 4. While the mean milk production per cow was lower with the FS1 farms compared to the FS2 and FS3 farms, milk production per cow was higher in FS2 farms compared to FS3 farms. In addition, there was no significant difference in mean EI between FS2 and FS3 farms. The purpose of partial confinement feeding with the FS3 farms was not collated as part of the research undertaken within Chapter 4. The purpose may have been to reduce pugging of pastures during the wet winter months or the delivery of by-products which would be impractical in a paddock, neither of which would necessarily automatically result in increased milk production per cow. In addition, the genetic merit of the FS2 herds may have been superior to the FS3 herds, illustrated by the higher average milk production per cow with the FS2 farms compared to the FS3 farms. Therefore, this remains an area for further exploration.

Fully confined feeding systems often result in reduced cow fertility, reduced longevity and varying health and welfare concerns (White *et al.*, 2002; Arnott *et al.*, 2015), which counteract the benefits of lowering EI with increased milk production per cow. Most importantly, breeding larger animals, with higher inputs from grains and other supplementary feeding, and increased collection of animal waste with partial/full confinement feeding, may contribute to a net increase in total farm GHG emissions. Increasing per cow milk production needs to be counter-balanced with a reduction in herd size, to produce similar total herd milk production, thus resulting in a reduction of total farm GHG emissions (Pryce and Bell, 2017). There is also an increasing societal demand that dairy cows have access to grazing pastures. This is evident in countries such as The Netherlands, where the dairy company FrieslandCampina promotes and rewards its dairy farmers with a premium price for milk from dairy cows that graze on pastures for a minimum of six hours per day for at least 120 days per annum (Elgersma, 2012). These specifications aim to preserve the natural image and contribute to the societal licence to produce food.

Another aspect to consider with increasing grain feeding on farm is the ethical concern as to whether this is the best use of the grain. World average per capita food availability for human consumption improved to 2,770 kilocalories/day in 2005/07, with the authors proposing that in theory, "there is sufficient global food consumption for nearly everyone to be well-fed" (Alexandratos and Bruinsma, 2012). Yet, this isn't occurring, with some 2.8 billion people living in poverty, defined as consuming < 2,500 kilocalories/day (Alexandratos and Bruinsma, 2012). Global estimates are that one-third of the annual cereal grain harvest is fed to livestock rather than for human consumption (Alexandratos and Bruinsma, 2012; Wilkinson and Lee, 2018). As the world's population continues to increase, the associated issues of human food demand and poverty is pertinent when considering feeding grain to livestock as an emissions mitigation option.

### 8.9 Consumer and supply chain response to mitigation

In the short to medium term, mitigation options that facilitate an increase in animal productivity, such as increased milk production per lactation, without compromising fertility and longevity, are likely to have the greatest on-farm adoption. Consumer acceptance will also be critical in driving practice change. Consumer pressure has already resulted in improved welfare standards for livestock through altering on-farm management practices. Some changes have been implemented through legislation, for example, the phasing out of sow stalls within the swine industry (Shields *et al.*, 2017). Other changes have been market-driven, for example, the increasing preference for non-caged hen egg production (Scrinis *et al.*, 2017).

For the dairy industry, feeding a plant-based source of dietary fat to reduce enteric $CH_4$ emissions appears socially acceptable. This is already occurring on some farms, for reasons not necessarily related to mitigation of GHG emissions (*e.g.* increasing CP in pelletised concentrates), with farmers not receiving formal recognition of this with an abatement credit. However, public acceptance of using synthetically-manufactured mitigation options, *e.g.* 3NOP, or the increase usage of antibiotics, *e.g.* ionophores, is currently questionable. In addition, it will be critical that there are no adverse effects of mitigation options on food safety, either real or perceived.

The use of DCD on pastures was banned in New Zealand in late 2012 when residues of the chemical was found in milk (Astley, 2013). While DCD is not considered a food safety risk, at the time there was no international standard for acceptable levels DCD in food, thus any trace of the chemical found in milk was considered to breach the declared Maximum Residue Limit under the *Codex Alimentarius* standards. The chemical was voluntarily removed from sale as the risk, in terms of jeopardising the New Zealand export market, was considered greater than the benefit to the industry (Astley, 2013). A process is currently underway to identify a threshold level of residues of compounds (like DCD and others) with a very low toxicology that could be introduced to the *Codex Alimentarius* (Eckard and Clark, 2018), after which DCD may once again be a viable mitigation option. However, given the previous consumer reaction to the discovery of DCD residues in milk, there may still be reservations about its re-introduction even when the international standards are modified.

The supply chain has already begun to place downward pressure on food industries to reduce their GHG footprint. An example includes the transnational consumer good company Unilever setting a target of reducing the GHG impact of their products by 50% by 2030 compared to a baseline of 2010, with 100% of their raw agricultural products farmed sustainably (Unilever, 2010). In 2013, the Australian dairy industry was the first in the world to be recognised as meeting Unilever’s Sustainable Agriculture Code (Unilever, 2013). For Australian dairy manufacturers, such as Fonterra, and farmers supplying product to these dairy manufacturing companies, demonstrating their GHG emissions credentials will become increasingly important for market access.

The development of CNBeef and CN30 programs for the red-meat industry of Brazil and Australia, respectively (Alves *et al.*, 2015; MLA, 2017a), is another example of market access from two of the world's largest meat exporters (MLA, 2017b). The programs aim to demonstrate their credentials of supplying a 'carbon neutral emission' product into the marketplace, with much of the neutrality of emissions proposed to be achieved through offsets with C sequestration in vegetation. Doran-Browne *et al.* (2018) illustrated that C neutrality of a wool, prime lamb and beef enterprise was achievable with stocking rates of up to 22 dry sheep equivalents/ha when > 20% of the farm enterprise was under tree cover over a 25-year period. Doran-Browne *et al.* (2018) did not undertake a full cost-benefit analysis of the implications of allotting such a large proportion of the farm to trees. This would be a critical component of the analysis in determining the success of allocating a proportion of dairy land to tree plantings for C sequestration to offset some of the farm's emissions.

## 8.10 Global carbon marketplace

A global marketplace, with a price premium placed on products with lower embedded GHG emissions, will become critical if global reduction in GHG emissions from agriculture are to be realised. The Genetic improvement mitigation option in the MACC analysis above was a good example of this, where the cost was found to be between +$16 and +$20/t $CO_2$e abated. A price premium paid for this milk could offset some or all of this cost to the farmer, thus becoming economically viable.

There are currently a range of C markets, including the Clean Development Mechanism provisions of the Kyoto Protocol, the European Union's Emissions Trading Scheme, California's cap-and-trade scheme, the Regional Greenhouse Gas Initiative (a group of 9 states in north-eastern USA), New Zealand's Emissions Trading Scheme, Australia's ERF and other voluntary markets (Newell *et al.*, 2013).

Most of these C markets operate as a cap-and-trade system, where a cap is set on the total amount of GHG emissions that can be emitted by companies, with the cap reduced over time. Trading brings flexibility that ensures emissions are reduced where it costs the least to do so (European Commission, 2018). The Clean Development Mechanism program is where Annex 1 developed world countries, with emission caps, undertake GHG emissions reduction projects in Annex 2 developing world countries, where there is no cap on emissions as part of the Kyoto Protocol. The developing world country receives certified emission reduction credits for the reduction in GHG emissions, which are purchased to offset capped emissions elsewhere (Newell *et al.*, 2013).

The Australian ERF works as a reverse auction where proponents estimate the emissions reduction achievable with a project (*e.g.* avoided deforestation) and the price that they are willing to accept for the emission reduction their project will deliver. The lowest bids are accepted by the Australian government, the proponent is allocated a credit for each t $CO_2$e of GHG abated and once the project is concluded, the proponent receives payment for the credits delivered at the price they bid at auction (DoEE, 2017). The most common ERF projects to date have involved vegetation activities, such as protecting native forests from land clearing, planting trees for C sequestration and regenerating native forest on previously cleared land. Currently there are two ERF methods applicable to the dairy industry; (1) Destruction of $CH_4$ generated from dairy manure in covered anaerobic ponds, and (2) Reducing GHG emissions by feeding dietary additives to milking cows. There has been zero adoption of either method, due in part to the need for ERF projects to meet the auction minimum target bid size of 2,000 t $CO_2$e/annum. Aggregation of multiple smaller projects (individual farms) is the only likely way of achieving this rate of abatement for much of the Australian dairy industry. Another limitation has been the cost of participation, reportedly as high as AU $100,000 over seven years for a large-scale northern Australian beef operation (Cohn, 2015).

To keep global warming below a 2°C increase, and preferably 1.5°C increase, agriculture will also need to contribute efforts to achieve a net neutral GHG emissions target. One of the concerns of applying a global C price on agricultural-related GHG emissions is that this could lead to impacts on food price, availability and security, especially in the more vulnerable regions of the developing world (Smith *et al.*, 2013; Herrero *et al.*, 2016). Some of the negative impacts of a C price on food could include the diversion of cropping land from food production to bioenergy or to C sequestration (Frank *et al.*, 2017). All this needs to be balanced against the reality that for much of the developing world, ruminant livestock represent much more than just a food source. Livestock have social and cultural significance, provide manure for heating, represent capital accumulation and provide draught power (Gerber *et al.*, 2013a).

Frank *et al.* (2018) suggests that as GHG-intensive livestock products such as ruminant meat becomes more expensive with increasing C prices, production expansion in the more extensive/rangeland pastoral regions of the world is disadvantaged, particularly in developing regions such as Latin America, South Asia and Sub Saharan Africa. This reduction/stagnation in expansion will need to be balanced against food security for these regions. As such, a high C price on food production will have a much greater impact on farmers from the developing world, relative to the developed world, given the significantly greater emissions per unit of product in these regions (Gerber *et al.*, 2013a). It is critical that through the design of C markets, to tackle global warming, we do not create a new class of poverty (Hasegawa *et al.*, 2015; Frank *et al.*, 2017). This is especially important given that the global poor or bottom 50% of income earners, with a daily income of < $2.97 purchasing power parity (PPP), are responsible for approximately 15% of global GHG emissions, while the top 10% of income earners with purchasing power parity > $23/day, comprising mainly the populations of developed countries such as Australia, USA, the European Union and Japan, are responsible for approximately 36% of global GHG emissions (Hubacek *et al.*, 2017).

Ultimately the C marketplace needs to be global in its intent and a key part of the global C market will be the consumer. The consumer is demanding more than just a low C footprint for their food. Farmers are constantly juggling a myriad of sustainability metrics, not just minimising the C footprint, but also including the need to maintain wildlife biodiversity and high animal welfare standards, improve water use efficiency while reducing water contamination of lakes and river streams, and minimising soil degradation and acidification (Tilman, 1999; Hockman *et al.*, 2013; Clark and Tilman, 2017). Therefore, an important consideration is whether the cost of implementing mitigation options to reduce on-farm GHG emissions should be the financial responsibility of the farmer. The implementation cost of many mitigation options has the potential to be greater than any associated increase in productivity (*e.g.* Moran *et al.*, 2011; Cotter *et al.*, 2015). In the developed, affluent world, the consumers of high-quality products, delivered through high standards of sustainability, will need to contribute to the mitigation implementation costs.

### 8.11 Emissions intensity versus net emissions reduction

Emissions intensity has been extensively and efficiently used to compare farm emissions and mitigation options. International supply chain markets have increasingly required a C footprint certification for imported products (Higgins *et al.*, 2015). For example, Australia was the first country in the world to comply with Unilever's Sustainable Agriculture Code, with all dairy production accredited by Dairy Australia meeting the demands of this code (Unilever, 2013).

A C footprint is an EI metric, based on GHG emissions per unit of product, rather than an absolute measure of GHG emissions. The trade-off between reducing EI and net emissions can be complex. Eckard *et al.* (2010), modelled the effect of low-quality versus high-quality diets for a pasture-based dairy system, adjusting stocking rates to achieve comparative pasture consumption per unit area. Under the high-quality pasture, the model predicted a 33% increase in stocking rate, resulting in a 38% increase in milk production (litres/ha). Emissions intensity (kg $CO_2$e/litre milk) was reduced by 19%, thus illustrating a win:win mitigation option in reducing EI, while improving productivity and thus profitability. However, net emissions increased by 26% due to the increased stocking rates. An alternative strategy may have been to increase diet quality but maintain similar milk production outputs from a smaller herd size and land mass with the residual land planted to trees for C sequestration. The alternative strategy would deliver both a reduction in EI and net emissions, an outcome that will become increasingly critical.

Most mitigation options explored in the MACC analysis delivered reductions in EI and total farm GHG emissions. While the options were not modelled together to ascertain the combined reduction in total farm GHG emissions, it was clear that reliance on these seven mitigation options, based on the assumptions followed here, could not deliver a net elimination of all on-farm GHG emissions. Reviews of technologies to reduce enteric $CH_4$ and $N_2O$ emissions from grazing-based dairy production systems have suggested that a net zero GHG-emissions Australian dairy industry is not currently feasible (Moate *et al.*, 2016; Eckard and Clark, 2018). While there are some promising mitigation opportunities being researched that have shown the potential to substantially reduce enteric $CH_4$ emissions, such as 3NOP and

*Asparagopsis*, this research is in its infancy. It remains unclear whether these will prove to be practical and acceptable methods for reducing enteric $CH_4$ emissions. Therefore, to meet our COP21 agreement obligations, a greater focus will be placed on reducing net emissions through sinks to offset the residual, unavoidable production emissions.

Dairying is reliant on well-fertilised, permanent pastures in high rainfall or irrigated regions. Thus, the scope for C sequestration in soils on long-established dairy farms is generally considered low compared to extensive livestock and cropping businesses (Cotching, 2012; Robertson *et al*., 2016). In addition, land values and productivity potential are also generally considered higher on dairying land than pastoral/cropping land and as such, there has traditionally been little economic incentive to reserve productive dairy land for tree plantings (Leddin *et al.*, 2012; Reisinger *et al.*, 2018), unless other co-benefits such as shelter, shade and improved property value have been achieved (DAFF, 2013). That said, a move towards GHG neutrality will provide a new incentive to find alternative sinks.

# CHAPTER 9 CONCLUSION

The overarching aims of this thesis were to estimate the GHG emissions of Australian dairy farms, using the Australian NGGI methodology, to compare with other pasture-based dairy systems throughout the world and then examine potential mitigation options to reduce GHG emissions attributed to milk production. To estimate GHG emissions, a localised focus within one region was explored (Tasmania) to represent similar geographic and climatic conditions. This was expanded to include modelling of the GHG emissions of dairy farms from throughout all dairying regions of Australia, reflecting the depth and breadth of the Australian industry. The results of these two analyses, in addition to the exploration of the influences of changing methodology on GHG emissions, confirmed that the EI of milk production of the Australian dairy industry is comparative to other pasture-based systems throughout the developed world.

Analysis of the Australian dataset highlighted the influence of FS on EI, with lower reliance on supplementary grain feeding, indicative of FS1 farms, increasing the EI of milk production. Therefore, a simplified approach to reducing the EI of Australian milk production might be to increase the intensification of FS, especially the FS1 farms, with increased grain feeding. However, this has far-reaching implications, beyond reducing dairy GHG emissions, which needs to be carefully balanced. Examples include dairy cattle consuming grain that may be better suited for human consumption or the need for increased consumer acceptance of milk harvested from farms that increase farm intensification, thus potentially reducing the time cows spend grazing pastures.

This thesis then examined a range of currently available mitigation options, using both dynamic and static modelling methods. Using DairyMod, a comparison of two methods of improving the NUE of the milking cow, to reduce $N_2O$ emissions, was undertaken. Reducing the N concentration of the diet to reduce the N source reduced $N_2O$ emissions by a magnitude greater than increasing the N concentration in milk, thus affirming the importance of improving the dietary balance for the milking cow in terms of her energy and protein intakes. This mitigation option is relevant for all dairy farms, especially where reliance on N-fertilised irrigated pastures can result in elevated dietary CP concentrations. Using the SGS Pasture Model, the importance of reducing the time between weaning and first mating for dairy heifer, thus allowing them to calve sooner and thus reduced her lifetime GHG emissions was shown. This has more application for the northern dairy industry where cows generally calve year-round, as opposed to the southern dairy industry where there is generally a condensed calving period, either once or twice per year, to match pasture supply.

The GHG-reduction potential of another five currently available mitigation options, in addition to an adaptation of the two above-mentioned options, were examined in COST, with the output results incorporated into a MACC analysis to compare the marginal cost of abatement against the reduction in EI. This was undertaken for four dairy farms, across contrasting FSs and varying EI profiles, to determine the magnitude of potential of each mitigation option. The overall conclusions from the MACC analysis was that all seven mitigation options reduced total farm GHG emissions, when estimated as the difference in EI between the baseline and strategy farm multiplied by milk production of the strategy farm. Some of the mitigation options also delivered an increase in milk production, thus win:win in terms of delivering a reduction in GHG emissions while improving farm productivity. This is a positive outcome to allow growth of the industry while remaining on a trajectory of reducing GHG emissions per unit of milk production.

Three of the seven mitigation options examined also delivered an increase in farm profitability across all four farms examined. The rate of abatement potential for some mitigation options reviewed was more reflective of the physical farm data rather than the FS. Thus, broad statements identifying clear pathways to reducing GHG emissions, based on FS, did not become evident. What was clearer was that reviewing inefficiencies on farm and implementing mitigation options that specifically target these inefficiencies, considering the whole farm system and economic implications of implementation, should be the priority of every farmer to deliver on-farm GHG emission reductions. It also must be noted that the results of the MACC analysis was a direct outcome of the input assumptions, with different input assumptions altering the GHG emission reduction potential of cost of implementation.

The scientific community will continue to identify and research mitigation options towards ‘creating’ a lower $CH_4$-emitting cow and improving N efficiency (animal, fertiliser and soil) while promoting management options to reduce on-farm wastage (*e.g.* more efficient energy consumption or improved herd longevity to reduce replacement rates and number of cows culled annually). The consumption of dairy products will continue to have an environmental footprint and as such, the supply chain has already begun to place downward pressure on the food industry to reduce its C footprint. Australian dairy farmers will increasingly be pressured to consider the GHG emissions implications of each management decision they make, identifying and implementing mitigation options that are additive and win:win for the farmer and the environment. The current focus of the Australian dairy industry has been to ascertain options to reduce the EI of milk production. While this has been useful to compare between farm systems, farm management and mitigation options, this has the potential to oversimplify solutions that do not necessarily result in a net reduction in total farm GHG emissions, a critical outcome to meet the Paris Agreement of zero net emissions, across the whole economy, by 2050. Mitigation options must be considered in the system and global context to ensure implementation does not inadvertently create a leakage of emissions to regions with higher C footprints, increase other sustainability issues or raise societal concerns. What the planet is experiencing is a net increase in total GHG emissions. Thus, we must find a balance between feeding an increasing global population while identifying and implementing cost-effective and management-ready GHG emissions reduction options to reduce individual farm, national and global dairy GHG emissions.

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Reference listing with bold first author (**Christie KM**) indicate publications submitted for the degree, reproduced in full in this thesis.

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## APPENDICES

List of Tables

**Table A4g** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Extended lactation mitigation option

# Appendix 1- Publication record

**Table A1a** Citations of the five published 1st author peer reviewed papers (totalling 66).

| Authors | Title | Journal | Year of publication | No. of citations |
|---|---|---|---|---|
| Christie KM, Rawnsley RP, Eckard RJ | A whole farm systems analysis of greenhouse gas emissions of 60 Tasmanian dairy farms | Animal Feed Science and Technology | 2011 | 37 |
| Christie KM, Gourley CJP, Rawnsley RP, Eckard RJ, Awty IM | Whole-farm systems analysis of Australian dairy farm greenhouse gas emissions | Animal Production Science | 2012 | 17 |
| Christie KM, Rawnsley RP, Harrison MT, Eckard RJ | Using a modelling approach to evaluate two options for improving animal nitrogen use efficiency and reducing nitrous oxide emissions on dairy farms in southern Australia | Animal Production Science | 2014 | 9 |
| Christie KM, Harrison MT, Trevaskis LM, Rawnsley RP, Eckard RJ | Modelling enteric methane abatement from earlier mating of dairy heifers in subtropical Australia by improving diet quality | Animal Production Science | 2016 | 2 |
| Christie KM, Rawnsley RP, Phelps C, Eckard RJ | Revised greenhouse-gas emissions from Australian dairy farms following application of updated methodology | Animal Production Science | 2018 | 1 |

Google Scholar, Web of Science or ResearchGate citations of journals, book chapters and conference proceedings as of 15 Feb 2019

In addition to the five 1st author peer reviewed papers, over the last eight years, I have also contributed to a range of other peer-reviewed papers, book chapters and conference proceedings researching GHG mitigation and climate change adaptation for Australia's livestock industries. These include:

- Rawnsley R, Dynes RA, **Christie KM**, Harrison MT, Doran-Browne NA, Vibart R, Eckard R (2018) A review of whole farm-system analysis in evaluating greenhouse-gas mitigation strategies from livestock production systems. *Animal Production Science* **58**, 980-989.
- Harrison MT, **Christie KM**, Rawnsley RP (2016) Assessing the reliability of dynamical and historical climate forecasts in simulating hindcast pasture growth rates. *Animal Production Science* **57**, 1525-1535
- Hayman G, **Christie K**, Harrison M, Cullen B, Ayre M, Armstrong D, Mason W, Rawnsley R, Nettle R, Beilin R, Waller S, Reichelt N, Brown R, White M, Phelps C (2016) Synergies and conflicts between adaptation and mitigation: future pathways for Australian dairy farms. *Proceedings of 6th Greenhouse Gas and Animal Agriculture Conference*, 14-18 February 2016, Melbourne, Victoria, Australia
- Hills J, McLaren D, **Christie K**, Rawnsley R, Taylor S (2016) Use of optical sensor to reduce nitrogen fertiliser inputs to intensively grazed pastures. *Proceedings of Precision Dairy Farming 2016*, 21-23 June 2016, Leeuwarden, The Netherlands.
- **Christie KM**, Doran-Browne NA, Rawnsley RP, Harrison MT, Eckard RJ (2014) A simple carbon-offset scenario tool (COST) for assessing sheep enterprise intervention options to reduce greenhouse gas emissions. *Proceedings of the 8th International Workshop on Modelling Nutrient Digestion and Utilisation in Farm Animals*, 15-17 September 2014, Cairns, Queensland, Australia.
- Cullen BR, Rawnsley RP, Eckard RJ, **Christie KM**, Bell MJ (2014) Use of modelling to identify perennial ryegrass plant traits for future warmer and drier climates. *Crop & Pasture Science* **65**, 758-766.

- Gollnow S, Lundie S, Moore AD, McLaren J, van Buuren N, Stahle P, **Christie K**, Thylmann D, Rehl T (2014) Carbon footprint of milk production from dairy cows in Australia. *International Dairy Journal* **37**, 31-38.
- Harrison MT, **Christie KM**, Rawnsley RP, Eckard RJ (2014) Modelling pasture management and livestock genotype interventions to improve whole-farm productivity and reduce greenhouse gas emissions intensities. *Animal Production Science* **54**, 2018-2028.
- **Christie KM**, Harrison MT, Rawnsley RP, Eckard RJ (2013) A simple Carbon Offset Scenario Tool for assessing dairy farm abatement options. *Proceedings of the 20th International Congress on Modelling and Simulation (MODSIM2013),* 1-6 December 2013, Adelaide, South Australia, Australia.
- Eckard R, Rawnsley R, Cullen B, Bell M, **Christie K** (2013) Modelling adaptation and mitigation strategies for southern livestock industries of Australia. *Proceedings of the 22nd International Grasslands Congress: Revitalising Grasslands to Sustain our Communities*, 15-19 September 2013, Sydney, New South Wales, Australia.
- Eckard R, Bell M, **Christie K**, Rawnsley R (2013) Chapter 11 Livestock. In ‘Living in a Warmer World: How a changing climate will affect our lives’ (Ed. J Salinger) pp 145-157. (CSIRO Publishing: Collingwood, Victoria, Australia).
- Lane P, Cullen B, Bell M, **Christie K**, Rawnsley R, Eckard R (2013) Chapter 9 Grasslands. In ‘Living in a Warmer World: How a changing climate will affect our lives’ (Ed. J Salinger) pp 120-133. (CSIRO Publishing: Collingwood, Victoria, Australia).
- **Christie KM**, Rawnsley RP, Smith RW (2012) Greenhouse gas emissions of lamb production in Tasmania. *Proceedings of the Climate Change Research Strategies for Primary Industries Conference,* 27-29 November 2012, Melbourne, Victoria, Australia.
- **Christie KM**, Rawnsley RP, Ball PD (2012) Frequency of wet and dry soil conditions in Tasmanian beef and sheep regions under future climate

scenarios. *Capturing Opportunities and Overcoming Obstacles in Australian Agronomy*, 14-18 October 2012, Armidale, New South Wales, Australia.

- **Christie K**, Rawnsley R, Cullen B, Bell M, Eckard R (2012) Predicting frequency of wet and dry soil conditions in Tasmanian dairy regions under future climate scenarios. *Capturing Opportunities and Overcoming Obstacles in Australian Agronomy*, 14-18 October 2012, Armidale, New South Wales, Australia.

I have been involved in many other activities relating to dairy GHG emissions. Key activities have included:

- Developed a new tool to estimate enteric $CH_4$ emissions for the Carbon Farming Initiative methodology "*Reducing greenhouse gas emissions in milking cows through feeding dietary additives*" in conjunction with the Australian Department of the Environment and Energy. As part of this process, I presented the tool at an Extension and Outreach Carbon Farming Futures Program training day to 60 people.
- On-going support to Dairy Australia, farmers and farm advisors in the use of the DGAS/ADCC. I have provided face-to-face and webinar training to > 200 people in the use of the calculator.
- Involved in several YouTube videos developed by Dairy Australia, summarising dairy GHG emissions (https://www.youtube.com/watch?v=OC1ZpYZNhjg&t=119s).
- Expert panellist in the upgrade of the methodology for estimating dairy GHG emissions in the National Greenhouse Gas Inventory in conjunction with the Australian Department of Environment.
- Working closely with Dairy Australia and external partners to incorporate the ADCC into DairyBase, including a presentation of the incorporation of the calculator at the Profitable Dairying in a Carbon Constrained Future Workshop.

- Collaboration with Fonterra NZ to compare the GHG emissions methodology of both countries' inventory to facilitate the estimation of the GHG emissions of their farmer base.

I have also been the first author or co-authored of published papers reviewing the role of N fertiliser on dairy farms. While not directly assessing the GHG emissions implications in these papers, N fertiliser is a source of $N_2O$ emissions, and therefore has relevance to the management of dairy farms and potential mitigation of these GHG emissions.

- **Christie KM**, Smith AP, Rawnsley RP, Harrison MT, Eckard RJ (2018) Simulated seasonal responses of grazed dairy pastures to nitrogen fertilizer in SE Australia: Pasture production. *Agricultural Systems* **166**, 36-47.
- Smith AP, **Christie KM**, Rawnsley RP, Eckard RJ (2018) Fertilizer strategies for improving nitrogen use efficiency in grazed dairy pastures. *Agricultural Systems* **165**, 274-282.
- **Christie K**, Rawnsley R, Eckard R (2018) Modelling nitrogen fertiliser by interaction for southern Australian dairy farms. In 'Proceedings of the 2018 Australasian Dairy Science Symposium', Palmerstone North, New Zealand.
- Rawnsley RP, Smith AP, **Christie KM**, Harrison MT, Eckard RJ (2019) Current and future direct of nitrogen fertiliser use in Australian grazing systems. *Crop & Pasture Science* (accepted for publication).
- **Christie KM**, Smith AP, Rawnsley RP, Harrison MT, Eckard RJ (2019) Simulated seasonal responses of grazed dairy pastures to nitrogen fertilizer in SE Australia: N losses and recovery. *Agricultural Systems* (in review).

## Appendix 2- Assumptions across all mitigation options

To undertake the MACC analysis, a series of assumptions were made across all mitigation options. These were:

- no alteration was made to any capital expenditure or management requirements with implement of each mitigation option;
- the cost of implementation reflects the difference between any expenditure associated with implementing the mitigation option (*e.g.* change in expenditure with purchasing a NI vs urea fertiliser) and change in income generated from implementing the mitigation option (*e.g.* increased or reduced milk income, reduced cost for supplementary feed);
- where the ME of the diet was altered with a mitigation option, each 5.5 MJ ME was converted into 1 litre milk (Moran, 2005);
- dietary mitigation options do not alter the milk composition (fat or protein content), only milk volume (litres) and thus kg FPCM for estimating EI of milk production;
- any change to milk production was valued at 45c/ litre ($6/ kg MS);
- no change in electricity consumption if annual milk production altered;
- where a change in diet results in the purchase of grain, fodder and/or by-products, the difference in pre-farm gate embedded emissions associated with these supplementary feeds are accounted for. For example, the Feeding fats mitigation option replaced some grain, with an embedded emission of 0.3 t $CO_2$e/t DM, with a by-product, with no C burden (GHG emissions were allocated to the primary product), thus reducing pre-farm gate embedded GHG emissions. As supplementary feed options were considered to displace each other, transport and associate GHG emissions were not included in the analysis (see section 2.5.2 of the need to consider transportation in mitigation analysis).

## Appendix 3- Implementation timeframe and assumptions

Some mitigation options were implemented immediately (first year) while others took up to six years to transition to full implementation. For these mitigation options (Genetic improvement, Reduced RR and Extended lactation), it was necessary to annualise the GHG emission reduction potential and cost of implementation, based on a series of assumptions and using a 5% discount rate for calculating the Net Present Value (cost of implementation and change in income from milk production). Each mitigation option also required a series of assumptions that were consistent across all four farms to estimate the change in GHG emissions, change in milk income and cost of implementation.

### Appendix 3a: Genetic improvement

The transition to full implementation for the Genetic improvement mitigation option for each stock class (milkers, heifers < 1 year of age and heifers > 1 year of age) varied each year based on the rate of implementation. For example, only the heifers born in Year 1 have a reduction in GHG emissions during Year 1.

The assumptions for this mitigation option were:

- 10% reduction in total enteric $CH_4$ across all stock classes (Huhtanen *et al.*, 2016; Danielsson *et al.*, 2017), although the mode of action was not defined (*e.g.* genetic differences between animals, differences in methanogenic population in animals, smaller rumen to increase passage rate);
- no change in annual total farm milk production;
- mitigation option was effective for 365 days per annum;
- cost of implementation was an additional $3 per milking cow during the transition phase.

**Table A3a** Proportion of stock class transitioned to genetically lower emitting animals at the end of each year of the 6-year transition period.

| Year of transition | Milking herd | Heifers < 1 year of age | Heifers > 1 year of age |
|---|---|---|---|
| Yr 1 | 0% | 100% | 0% |
| Yr 2 | 0% | 100% | 100% |
| Yr 3 | 20% | 100% | 100% |
| Yr 4 | 40% | 100% | 100% |
| Yr 5 | 70% | 100% | 100% |
| Yr 6 | 100% | 100% | 100% |
| Annualised | 38.3% | 100% | 83.3% |

**Appendix 3b: Feeding fats**

The assumptions for this mitigation option were:

- replace half of the baseline grain fed to the milker during summer and autumn with a new high fat supplement;
- the baseline grain composition (DMD, CP and fat %) was based on each farm's lab-analysed feed quality data during summer and autumn (Chapter 4 dataset);
- the high fat supplement had a DMD of 75%, CP of 20% and fat of 10% (based on a combination of several high fat feed such as grape marc, dried distillers grain and hominy);
- the high fat supplement was delivered through the milking shed for 130 days per annum;
- potential for altered milk production if the DMD of the diet changed;
- the baseline grain cost $300/t DM (long-term average for Australia; Dairy Australia hay and grain reports; https://www.dairyaustralia.com.au/industry/farm-inputs-and-costs/hay-and-grain-report-overview) and the new high fat supplement cost $360/t DM (estimate).

### Appendix 3c: Improved DMD to CP ratio

The assumptions for this mitigation option were:

- replace half of the baseline grain (DMD 82% and CP 13%) fed to the milking herd with maize silage (DMD of 68% and CP of 8%), with the new grain/maize silage supplement having a DMD of 75% and CP of 10.5%;
- potential for altered milk production if the DMD of the diet changed;
- baseline grain cost $300/t DM (long-term average for Australia; Dairy Australia hay and grain reports) and maize silage cost $220/t DM (assumed to be grown/harvested on a neighbouring farm and purchased by the dairy farm; https://www.pioneerseeds.com.au/corn-silage/product-information/silage-technical-insights/maize-product-options.html);
- for the 30% wastage of the maize silage when fed in the paddock for FS1 farms, replaced each kg DM of grain with 1.3 kg DM silage to incorporate the 30% wastage, thus increasing the cost of the maize silage to $286/t DM and overall grain/ maize silage supplement to $293/t DM as fed;
- for the 10% waste of the maize silage when fed on a feedpad for the FS2/3 farms, replaced each kg DO of grain with 1.1 kg DM silage to incorporate the 10% wastage, thus increasing the cost of the maize silage to $242/t DM and overall grain/maize silage supplement to $271/t DM as fed;
- no change in on-farm waste emission factors (*e.g.* integrated MCF) when cows are fed silage on a feedpad as opposed to in the paddock. This would need to be reviewed for individual farms, based on their current waste management practices and thus beyond the scope of this analysis.

### Appendix 3d: NI- N fertiliser

The assumptions for this mitigation option were:

- NI applied as a coating on N fertiliser;
- 60% of the annual purchased N fertiliser was coated with a NI;
- all fertiliser applied during the wetter periods of the year when $N_2O$ losses were greatest was coated with a NI (autumn through spring);

- efficacy of the NI was a 30% reduction in N-fertiliser related $N_2O$ emissions (Kelly *et al.*, 2008; Suter *et al.*, 2016);
- cost of the NI coated fertiliser was an additional \$200/t relative to uncoated fertiliser (urea \$1300/ t N and NI \$1500/ t N; Eckard (2019) pers. comm.);
- no change in cost of spreading the NI coated fertiliser onto the paddock;
- no additional pasture production, and thus no change in milk production, due to inconsistencies in published research (Dougherty *et al.*, 2016; Suter *et al.*, 2016).

**Appendix 3e: NI- urinary deposition**

The assumptions for this mitigation option were:

- NI applied as a liquid spray onto pastures grazed by the milking herd;
- milking herd spend 15% of their time in laneways and dairy shed, thus 85% of urine and dung is deposited onto pastures for all four farms;
- review of FS2/3 farms also spending an additional 5% of their time on a feedpad, thus 80% of urine and dung is deposited onto pastures;
- NI was sprayed onto pastures during the wetter periods of the year when $N_2O$ losses were greatest, and thus assumed to be applied for 180 days per annum (autumn through spring);
- efficacy of the NI was a 30% reduction in urinary-N related $N_2O$ emissions to compare with the NI- N fertiliser option (Kelly *et al.*¸ 2008);
- cost of NI was AUD \$24/ha.annum (based on 15€/ha and exchange rate of AUD \$1.60/€; Macleod *et al.* 2015);
- additional cost of applying the NI of \$1,800/annum (based on 0.5 hrs/day for 180 days and labour at \$20/hr).

**Appendix 3f: Reducing RR**

The transition to full implementation for the Reduced RR mitigation option took six years to implement with a smaller reduction in replacement rate the first two years, with the number of heifers no longer retained varying between farms depending on their baseline heifer retainment rate.

The assumptions for this mitigation option were:

- reduce the replacement rate by 20% by the end of the six-year period;
- raising fewer heifers did not alter the diet quality or quantity for the remaining replacement animals;
- cost to raise a heifer from birth to calving estimated at $1000/head.

**Table A3b** Proportional reduction in replacement heifers at the end of each year of the 6-year transition period and number of heifers no longer retained each year.

| Year of transition | % of heifers no longer retained | FS 1 High EI | FS1 Low EI | FS2/3 High EI | FS2/3 Low EI |
|---|---|---|---|---|---|
| Yr 1 | 10% | 1 | 1 | 1 | 4 |
| Yr 2 | 20% | 2 | 2 | 3 | 8 |
| Yr 3 | 40% | 4 | 4 | 6 | 15 |
| Yr 4 | 60% | 7 | 6 | 8 | 23 |
| Yr 5 | 80% | 9 | 7 | 11 | 30 |
| Yr 6 | 100% | 11 | 9 | 14 | 38 |
| Annualised | 52% | 6 | 4 | 7 | 20 |

**Appendix 3g: Extended lactation**

The transition to full implementation for the Reduced RR mitigation option took six years to implement, with half of the milking herd transitioned to an extended lactation by the end of the 3rd year, with the other half of the herd fully transitioned by the end of the 6th year.

The assumptions for this mitigation option were:

- compare six 12-month calving intervals (baseline farm; 300 days lactating and 65 days non-lactating) versus four 18-month calving intervals (mitigation option; 482 days lactating and 65 days non-lactating) so same number of days present on the farm before culling;

- extended lactation cows produced 95% of the daily milk production rate of the baseline lactation cows (Auldist *et al.*, 2007);
- extended lactation cows required additional purchased grain (0.125 t DM/cow.annum at $300/ t DM) and fodder (0.1 t DM/cow.annum at $150/t DM) (Browne *et al.*, 2015);
- no change in management expenses associated with extended lactation based on lower labour requirements for calf rearing lower health and breeding-related expenses offsetting increased labour and energy costs with milking (Abdelsayed *et al.*, 2015);
- effect of extended lactation on replacement heifers was not included in the analysis (see Chapter 8.6.5 for rationale for no change in heifer numbers),
- half of the milking herd transitioned to extended lactation by the end of the 3$^{rd}$ year, with the other half of the herd fully transitioned by the end of the 6$^{th}$ year. Therefore the annualised average change in GHG emissions, supplementary feeding and milk production reflects half of total change in emissions, supplementary feeding and milk production at the end of the six-year transition.

# Appendix 4- Results of the MACC analysis

## Appendix 4a: Genetic improvement mitigation option

**Table A4a** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Genetic improvement mitigation option.

| | **FS1 High EI** | **FS1 Low EI** | **FS2/3 High EI** | **FS2/3 Low EI** |
|---|---|---|---|---|
| Baseline GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions (t $CO_2$e/annum) | 1,541 | 1,334 | 2,328 | 4,839 |
| Strategy milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Strategy EI (kg $CO_2$e/kg FPCM) | 1.19 | 1.08 | 1.05 | 0.97 |
| Reduction in EI (kg $CO_2$e/kg FPCM) | 0.033 | 0.028 | 0.024 | 0.026 |
| Reduction in total farm GHG emissions (t $CO_2$e/annum) | 42.1 | 34.8 | 53.1 | 127.6 |
| Net cost of implementation ($/annum)[1] | $857 | $660 | $829 | $2,130 |
| Cost effectiveness ($/t $CO_2$e reduction)[1] | +$20 | +$19 | +$16 | +$16 |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

[1] Cost of implementation and cost-effectiveness based on 5% discount on the Net Present Value

## Appendix 4b: Feeding fats mitigation option

**Table A4b** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Feeding fats mitigation option.

| | FS1 High EI | FS1 Low EI | FS2/3 High EI | FS2/3 Low EI |
|---|---|---|---|---|
| Baseline GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions (t $CO_2$e/annum) | 1,547 | 1,355 | 2,315 | 4,819 |
| Strategy milk production (t FPCM) | 1,316 | 1,239 | 2,262 | 5,078 |
| Strategy EI (kg $CO_2$e/kg FPCM) | 1.18 | 1.09 | 1.02 | 0.95 |
| Reduction in EI (kg $CO_2$e/kg FPCM) | 0.048 | 0.019 | 0.047 | 0.046 |
| Reduction in total farm GHG emissions (t $CO_2$e/annum) | 63 | 24 | 107 | 233 |
| Net cost of implementation ($/annum) | -$4,102 | -$1,776 | -$7,666 | -$17,028 |
| Cost effectiveness ($/t $CO_2$e reduction) | -$65 | -$74 | -$71 | -$73 |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

**Appendix 4c: Improved DMD to CP ratio**

**Table A4c** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Improved DMD to CP ratio mitigation option.

| | **FS1 High EI** | **FS1 Low EI** | **FS2/3 High EI** | **FS2/3 Low EI** |
|---|---|---|---|---|
| Baseline GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions (t $CO_2$e/annum)[2] | 1,572 | 1,360 | 2,337 (2,317) | 4,936 (4,911) |
| Strategy milk production (t FPCM)[2] | 1,289 | 1,229 | 2,197 (2,197) | 4,947 (4,947) |
| Strategy EI (kg $CO_2$e/kg FPCM)[2] | 1.22 | 1.11 | 1.06 (1.05) | 1.00 (0.99) |
| Reduction in EI (kg $CO_2$e/kg FPCM)[2] | 0.004 | 0.007 | 0.007 (0.016) | -0.003 (0.002) |
| Reduction in total farm GHG emissions (t $CO_2$e/annum)[2] | 5 | 9 | 15 (36) | -13 (12) |
| Net cost of implementation ($/annum)[2] | $628 | -$1,061 | $6,352 (-$11,831) | $12,762 (-$9,262) |
| Cost effectiveness ($/t $CO_2$e reduction)[2] | +$135 | -$125 | +$415 (-$329) | +$980 (-$761) |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

[2] Values in brackets for FS2/3 farms denoting the results of when maize silage wastage was reduced from 30% to 10%.

## Appendix 4d: NI- N fertiliser mitigation option

**Table A4d** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the NI- N fertiliser mitigation option.

| | **FS1 High EI** | **FS1 Low EI** | **FS2/3 High EI** | **FS2/3 Low EI** |
|---|---|---|---|---|
| Baseline GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions (t $CO_2$e/annum) | 1,576 | 1,358 | 2,366 | 4,959 |
| Strategy milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Strategy EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Reduction in EI (kg $CO_2$e/kg FPCM) | 0.006 | 0.008 | 0.007 | 0.002 |
| Reduction in total farm GHG emissions (t $CO_2$e/annum) | 8 | 10 | 16 | 8 |
| Net cost of implementation ($/annum) | $1,728 | $2,292 | $3,600 | $1,896 |
| Cost effectiveness ($/t $CO_2$e reduction) | +$228 | +$228 | +$228 | +$228 |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

## Appendix 4e: NI-urinary deposition mitigation option

**Table A4e** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the NI-urinary deposition mitigation option.

| | FS1 High EI | FS1 Low EI | FS2/3 High EI | FS2/3 Low EI |
|---|---|---|---|---|
| Baseline GHG emissions ($t\ CO_2e$/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2e$/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions ($t\ CO_2e$/annum) [3] | 1,519 | 1,308 | 2,292 (2,297) | 4,801 (4,811) |
| Strategy milk production (t FPCM) [3] | 1,294 | 1,230 | 2,224 (2,224) | 4,992 (4,992) |
| Strategy EI (kg $CO_2e$/kg FPCM) [3] | 1.17 | 1.06 | 1.03 (1.03) | 0.96 (0.96) |
| Reduction in EI (kg $CO_2e$/kg FPCM)[3] | 0.050 | 0.049 | 0.040 (0.038) | 0.033 (0.031) |
| Reduction in total farm GHG emissions ($t\ CO_2e$/annum) [3] | 64 | 60 | 90 (85) | 166 (156) |
| Net cost of implementation ($/annum)[3] | $5,904 | $4,032 | $7,104 | $7,464 |
| Cost effectiveness ($/t $CO_2e$ reduction)[3] | +$92 | +$67 | +$79 (+$84) | +$45 (+$47) |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

[3] Values in brackets for FS2/3 farms denoting the results of when the cows spent an additional 5% of their time on a feedpad as opposed to grazing.

**Appendix 4f: Reduced RR mitigation option**

**Table A4f** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Reduced RR mitigation option.

| | FS1 High EI | FS1 Low EI | FS2/3 High EI | FS2/3 Low EI |
|---|---|---|---|---|
| Baseline GHG emissions (t $CO_2$e/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI (kg $CO_2$e/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions (t $CO_2$e/annum) | 1,572 | 1,359 | 2,368 | 4,929 |
| Strategy milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Strategy EI (kg $CO_2$e/kg FPCM) | 1.21 | 1.11 | 1.06 | 0.99 |
| Reduction in EI (kg $CO_2$e/kg FPCM) | 0.008 | 0.007 | 0.006 | 0.008 |
| Reduction in total farm GHG emissions (t $CO_2$e/annum) | 11 | 9 | 14 | 38 |
| Net cost of implementation ($/annum)[1] | -$4,540 | -$3,893 | -$5,751 | -$15,801 |
| Cost effectiveness ($/t $CO_2$e reduction)[1] | -$422 | -$425 | -$419 | -$420 |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

[1] Cost of implementation and cost-effectiveness based on 5% discount on the Net Present Value

## Appendix 4g: Extended lactation mitigation option

**Table A4g.** Summary of the change in on-farm and pre-farm gate embedded emissions, change in milk production and milk income, reduction in emissions intensity of milk production, cost of implementation and cost-effectiveness on an annualised basis for the Extended lactation mitigation option.

| | **FS1 High EI** | **FS1 Low EI** | **FS2/3 High EI** | **FS2/3 Low EI** |
|---|---|---|---|---|
| Baseline GHG emissions ($t CO_2e$/annum) | 1,583 | 1,369 | 2,382 | 4,967 |
| Baseline milk production (t FPCM) | 1,294 | 1,230 | 2,224 | 4,992 |
| Baseline EI ($kg CO_2e$/kg FPCM) | 1.22 | 1.11 | 1.07 | 1.00 |
| Strategy GHG emissions ($t CO_2e$/annum) | 1,600 | 1,381 | 2,398 | 5,007 |
| Strategy milk production (t FPCM) | 1,339 | 1,265 | 2,269 | 5,105 |
| Strategy EI ($kg CO_2e$/kg FPCM) | 1.19 | 1.09 | 1.06 | 0.98 |
| Reduction in EI ($kg CO_2e$/kg FPCM) | 0.028 | 0.021 | 0.014 | 0.014 |
| Reduction in total farm GHG emissions ($t CO_2e$/annum) | 38 | 27 | 31 | 72 |
| Net cost of implementation ($/annum)[1] | -$11,529 | -$9,306 | -$11,599 | -$29,693 |
| Cost effectiveness ($/t $CO_2e$ reduction)[1] | -$305 | -$343 | -$370 | -$413 |

NOTE: a negative cost of implementation or cost-effectiveness indicates a profit is realised with the implementation of the mitigation option, relative to the baseline farm system.

[1] Cost of implementation and cost-effectiveness based on 5% discount on the Net Present Value